IMPACT OF ARTIFICIAL AERATION ON NUTRIENTS IN A SMALL EUTROPHIC LAKE

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Impact of Artificial Aeration on Nutrients in

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The Supervisory Committee certifies that this ***disquisition*** complies with North Dakota State University's regulations and meets the accepted standards for the degree of

DOCTOR OF PHILOSOPHY

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ABSTRACT

The Heinrich- Martin Dam Impoundment (HMDI), located in northcentral LaMoure County, North Dakota, is an important water body for fishing, boating, and other recreational activities. To eliminate the thermal stratification and low dissolved oxygen (DO) zone near the bottom, an artificial aeration system was installed and operated by the North Dakota Game and Fish Department (NDG&F). This study was conducted to investigate whether aeration improved water quality of HMDI and to evaluate aeration as a tool to effectively manage eutrophication in lakes of this type.

Field monitoring and laboratory analyses were conducted during three consecutive summers, in 2010, 2011 and 2012, to evaluate the impact of aeration on the spatial and temporal variation of physical, chemical, and biological water quality. Variables monitored included total and dissolved forms of inorganic nitrogen (N) and phosphorus (P), chlorophyll *a*, turbidity, water temperature and dissolved oxygen (DO). Field sampling was carried out during aerated conditions in 2010 and 2012 and under non-aerated conditions in 2011.

The study revealed no significant differences between aeration and non-aeration conditions in soluble reactive phosphorus (SRP), total phosphorus (TP), or total nitrogen (TN) concentrations in the water column. In contrast, mean ammonia-nitrogen concentration at the near-bottom layers during non-aerated conditions decreased significantly under artificial aeration, while mean nitrate and nitrite concentrations increased significantly under aeration. Under aerated conditions, dissolved inorganic nutrients, TN, TP, temperature, and algae were homogenously distributed throughout the water column. Aeration expanded aerobic habitats for fish and distributed bio-available nutrients, stimulating algal growth throughout the water column. These results indicate that the existing aeration system vertically mixed nutrients

throughout the water column of the HMDI. Chlorophyll *a* results showed that aeration distributed algae throughout the water column and circulated available nutrients for their growth. In addition, the results indicated that aeration improved water quality as measured by Secchi depth, turbidity, DO and algal biomass based on recommended levels by NDDoH. In a nitrogen-limited, phosphorus-rich water body, like HMDI, lowering phosphorus (P) load rather than nitrogen (N) load is recommended as a means of reducing algal biomass.

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DEDICATION

This dissertation is dedicated to

My parents, Punchimahathaya and Rosalin Balangoda

My husband, Malinda Vikum Sanjaka

and

Our son, Mihin Vimukthi Benaragama

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CHAPTER 1. INTRODUCTION

1.1. Introduction

Natural eutrophication is an aging process of lakes and reservoirs. This typically slow process can be greatly accelerated by human interventions due to addition of excess nutrients from a variety of sources. This accelerated process is called cultural eutrophication. Excess nutrients of concern are nitrogen (N) and phosphorous (P), and sources of these nutrients include both point source (PS) and non-point sources (NPS), such as industrial waste water, agricultural runoff and urban runoff. Non-point source loads are mostly driven by rainfall events; however, PS loads are continuous discharges from a source, such as waste water effluent from treatment plant leachate from waste disposal sites (Carpenter et al., 1998; Matlock et al., 2013). In addition to external supplies of nutrients, a water body also receives nutrients from internal dynamics; for instance, nutrient flux from the sediment bed, sediment resuspension and bioturbation also contribute to eutrophication of lakes.

The nutrient enrichment can lead to a wide variety of problems, including excess algal growth; toxin production by some cyanobacteria species (for example: *Anabaena* sp. and *Planktothrix* sp.) (Anderson et al, 2002); shifts in phytoplankton species composition; diurnal dissolved oxygen (DO) changes; low DO, especially near the bottom of the water body; increase in turbidity; shifts in fish species composition towards less desirable species and reduction of aesthetic value (Raven and Geider., 1988). Among these consequences, the most conspicuous effect of eutrophication is the formation of noxious foul-smelling phytoplankton blooms (Chislock et al., 2013), which may severely deplete DO and create anoxic zone near the bottom. The consequences of these blooms are serious water quality degradation because the impacts can be tangible, such as fish kills and strong odors (Carpenter et al., 1998; Smith, 1999).

Dissolved oxygen concentration, water temperature, algal biomass and turbidity and nutrient concentration in a water body depend upon each other, so when changing one parameter it affects directly or indirectly on other parameters. The DO concentration is one of the most important factors for determining survival of aquatic organisms and balance of the water body. In a highly productive lake, aquatic organisms experience supersaturation of DO during daytime due to oxygen production by photosynthesis. During nighttime, due to a high demand for oxygen consumption, organisms experience undersaturation. So depletion of DO may cause sudden and complete fish kill in nutrient rich eutrophic lakes as a consequence of algae-mediated de-oxygenation. Further, DO solubility depends upon the water temperature. The increasing temperature during summer decreases DO solubility in water more than in colder months. Moreover, biochemical and physiological processes are governed by temperature; for instance, aquatic organisms have preferred ranges of temperature (for cold water fish 13-21°C; for warm water fish 22.8-30°C) (Stiekney 2000 and Ficke et al. 2005) and microbial decomposition of organic matter enhances in warm conditions more than in cold conditions. Further, algal growth rate is controlled by temperature and after optimum temperature, algal growth is inhibited with further increasing temperature (Raven and Geider, 1988; Cassidy, 2011). In addition, different algal groups (blue-green algae, green algae and diatoms) need different range of temperature; therefore, seasonal variation of algal species is possible in response to different water temperatures according to season (Cassidy, 2011).

In addition to water temperature, algal growth depends upon other environmental conditions, such as amount of light and nutrient level. Algae consume nutrients in a fixed stoichiometric ratio (dry-weight mass ratio for N and P is 7.2:1); therefore, if limiting nutrient either N or P is depleted, the eutrophication process may be reversed due to the limited supply of

least available nutrient (Hecky et al., 1993). The availability of dissolved inorganic N and P is controlled by presence of DO. For instance, under oxic conditions stimulation of microbial decomposition of organic matter releases nutrients into the water column, while under anoxic conditions, denitrification may serve as a main mechanism of removing bioavailable nitrate-nitrogen from the water column. Moreover, dissolved inorganic phosphorus released by Fe-desorption which is controlled by environmental conditions, and nutrient concentration variations eventually affect eutrophication of lakes. In addition to nutrients, algal growth can be limited due to light availability, which is needed for their primary production. Light can be limited due to algal growth itself or by other suspended solids.

Eutrophication of lakes and reservoirs is a major concern in North Dakota (ND). The North Dakota Department of Health (NDDoH) water quality assessment reports showed that the main causes of aquatic life impairment in lakes and reservoirs were likely due to low DO resulting from eutrophication (NDDoH, 2010 & 2012). According to the reports, major sources of nutrient loading are erosion, runoff from cropland, runoff from animal feeding operations and hydrologic modifications (wetland drainage, channelization and ditching), which result in algal bloom and noxious aquatic plant growth (NDDoH, 2012). The potential economic losses due to eutrophication in the United States freshwaters are approximately $2.2 billion annually, and the economic losses were attributed to loss of biodiversity (eutrophication decreases diversity and richness of aquatic macroinvertebrtaes, fish, and aquatic primary producers) (Dodds et al, 2009), decrease lake front property values and increase recreational costs (Dodds et al., 2009).

Due to increased demand of freshwater resources, protecting diminishing water quality is becoming the most pressing environmental issue. Several management strategies have been practiced to control the eutrophication in lakes and reservoirs, such as point and non-point source

control techniques, internal source control techniques, and ecosystem recovery techniques (Sondergaard et al., 2000; Ha et al., 2013). Point and non-point sources may be regulated by state water quality agency, environmental protection agency (EPA) and best management practices (BMPs). Ecosystem recovery techniques, such as biomanipulation and macrophyte implantation are low investment (Ha et al., 2013; Angeler et al., 2003); however, it takes a long time to restore eutrophic lakes (Ha et al., 2013). Another consideration is that even when the external nutrient loading is controlled, still lakes can maintain their trophic status due to internal processes (sediment resuspension, nutrients flux from the sediment bed, bioturbation, and nitrogen fixation); therefore, it is crucial to control the internal source of nutrients for effective lake restoration (Lauridsen et al., 2003). Different techniques, such as sediment dredging, nutrient precipitation, and physical mixing as artificial aeration could be used to control nutrient release from internal processes (Liboriussen et al., 2009; Soltero et al., 1994; Prepas and Burke 1997).

Among those internal source control techniques, artificial aeration, or oxygenation, is one of the frequently used techniques in the early 21st century for lake destratification (Cooke et al., 2005). The intention of this technique is to eliminate thermal stratification through either mechanical pumping or by diffused air pumping, to increase circulation within lakes and to increase dissolved oxygen in the near-bottom water for sustaining the fish population (Beutel, 2002). Studies have yielded different conclusions on N and P concentrations in the overlying water column under the conditions of aeration (Ashley, 1983; Osgood and Stiegler, 1990; Gachter and Wehrli, 1998; Hanson and Austin, 2012). For example, internal P cycling was not affected by increased DO in the near-bottom water (Gachter and Wehrli, 1998); nutrient concentration in the overlying water column depends upon external runoff (Ashley, 1983; Osgood and Stiegler, 1990) and construction projects in the watershed (Hanson and Austin,

2012); partial aeration (Grochowska and Gawronska, 2004); and no changes of total nitrogen (TN) and total phosphorous (TP) concentration due to artificial aeration (Jungo et al., 2001). All these studies imply that there is no solid conclusion about whether artificial aeration increases or decreases dissolved inorganic N and P in the overlying water column under the conditions of aeration. Therefore, better understanding the effects of artificial aeration on nutrient concentration in the water column is crucial for effective lake management.

In addition, understanding the impacts of aeration on the entire water body will provide further information for lake managers to determine whether the water body is completely destratified or not. Therefore, collecting samples throughout the water body will provide more information on water quality changes in the whole water body under the condition of aeration. Many studies employed only sampling of selected parts of their whole study area (Ashley, 1983; Soltero et al., 1994; Prepas and Burke, 1997; Heo and Kim, 2004). In addition, until now, no study has carried out water quality monitoring designed specifically to maximize obtaining information for studying the impact of aeration on dissolved N and P concentrations representative of the whole study area, including areas near and far from the aeration systems, across the water columns in deep and shallow areas, and other critical areas. Identifying impacts of aeration on dissolved N and P concentration in the overlying water column in the entire lake and its subsequent effects on algal growth is essential to study for effective lake restoration and to control algal blooms during summer.

This study was carried out at the Heinrich Martin Dam Impoundment (HMDI), which is a small, eutrophic reservoir used for fishing, boating and other recreational purposes. There was a previous study carried out in 2008 at the HMDI to identify causes for low DO in the water column due to experience of anoxia in near-bottom water during summer. Further, the HMDI is

under the supervision of North Dakota Game and Fish Department (NDG&F) which maintains the water quality according to the NDDoH recommended water quality criteria for class 3 lakes and reservoirs (maintains for warm water fishery). The NDG&F installed an aeration system (air diffuser system) in 2006 at the HMDI to increase DO concentration in near- bottom water at the deepest area of the HMDI, and observed improvement of DO in the water column. The NDG&F, however, required study of any consequences attributable to existing aeration system on nutrient concentrations in the water column, which directly affect eutrophication of the impoundment. Therefore, whether the existing aeration system suppresses nutrients in the water column along with other water quality parameters was expected to be studied to evaluate the aeration system control of the eutrophication at the HMDI. So, this study was initiated to assess water quality variations under aerated conditions and to provide recommendations and/or alternative methods to the NDG&F to control the eutrophication at the HMDI.

1.2. Research Problem Statement

Mortimer (1941) studied the relationship between P and oxygen concentration in the near-bottom waters and he found that the input of oxygen into the lake could be an effective method to restore the eutrophic lakes. Increased DO can prevent the accumulation of odor-forming gases like hydrogen sulfide (H_2S), shorten the anoxia period in the near-bottom water and increase the redox potential. Since 1941, numerous field and laboratory studies have indicated that lakes subjected to artificial aeration show improved DO concentration in the hypolimnion (Ashley, 1983; Soltero et al., 1994; Liboriussen et al., 2009). In addition to improved DO concentration in the near-bottom waters by artificial aeration, numerous studies, including both field and laboratory studies described in the literature, have aimed to measure N and P concentrations in the overlying water column under the conditions of aeration (Holdern

and Armstrong, 1980; Eckert et al., 1997; Moore et al., 1998; Fisher et al., 2005), but to date, results have been inconclusive in regard to the effectiveness of aeration.

Laboratory experiments showed that in oxic conditions, P is chemically bound to insoluble Fe (III) compounds, while under anoxic conditions Fe (III) is reduced to soluble Fe (II) compounds and subsequently both iron and chemically bound phosphates return to the overlying water column. However, under laboratory-based conditions, when sediment cores and water are transferred to the laboratory, their characteristics change and results may not be applicable to field conditions. For example, when transferring sediment cores, a formerly reduced sediment layer exposed to oxidation has a higher P binding capacity in the upper most sediment layer (Hietanen and Lukkari, 2007). Therefore, this uppermost layer can act as a barrier to P release from deeper parts of the sediment layers. Overall, laboratory-based results may not be useful for effective management of real systems.

Studies have shown that the impacts of aeration on water quality can be masked by other factors, such as external runoff, construction projects in watershed during study period and the combination of other lake restoration methods used with aeration (Soltero et al., 1994; Prepas and Burke, 1997; Heo and Kim, 2004). A long-term study of artificial mixing in two eutrophic Swiss lakes has shown that internal P cycling was not affected by increased hypolimnetic DO concentration; however, an overall reduction in P concentration was observed and found to be caused by temporal changes in external loading during the study (Ashley, 1983; Osgood and Stiegler, 1990; Gachter and Wehrli, 1998). Additionally, reduction of P content may also be due to the precipitation of released phosphates under anoxic conditions prior to the start of aeration. Therefore, their results cannot be categorically attributed to impacts of aeration on P release. Further, Hanson and Austin (2012) studied the effects of destratification on water quality from

1973 to 2010 under both aerated and non-aerated conditions at Sweeney Lake, Minnesota. However, they mentioned that their results were influenced by from storm water inputs, and the lake water budget is mainly comprises runoff from highways and roads. In addition, there were major construction projects during the study period, and storm water runoff was a major concern; therefore, effects of aeration on water quality could be masked by other environmental conditions and interannual variability. Thus, understanding the impacts on water quality from aeration will help to design effective lake restoration methods for eutrophic lakes.

In many instances, a partial sampling plan does not represent impacts of aeration on the whole study site, and partial aeration caused incomplete destratification that was responsible for undesirable water quality results (Ashley, 1983; Cowell et al., 1987; Soltero et al., 1994; Prepas and Burke, 1997; Heo and Kim, 2004). In many studies, researchers collected samples only from the deepest location and closer to the aerator (Ashley, 1983; Soltero et al., 1994; Prepas and Burke, 1997; Heo and Kim, 2004) and concluded that aeration decreased summer hypolimnetic orthophosphates, total phosphorous (TP), and hypolimnetic ammonium concentrations. In such cases, N and P concentrations for the entire lake cannot be determined based on this limited sampling. Additionally, another study carried out at Lake Brooker in northwest Hillsborough County, Florida, showed that annual mean TP and SRP concentrations decreased from June, 1981 to May, 1982 under destratified conditions and increased these concentration from June, 1982 to June, 1983 when aeration was in progress (Cowell et al., 1987); however, in that case samples were collected only at two locations, which were located at opposite ends of the lake throughout the three years study period (15 months pre aeration and 2 years during aeration). So, variation of nutrient concentrations for the entire lake was not represented by their results. Therefore, limited studies have been done on monitoring water quality and sample analyses

across the depths from surface to bottom and across the sampling locations, including deep and shallow locations and closer to the aeration system as well as away from the aeration system. Water quality monitoring and sampling analyses on the whole sampling site will be crucial to lake managers for understanding the impacts of aeration on the whole lake and to improve the existing aeration system.

Studies have yielded variable conclusions about effects of lake aeration (mixing) on N and P release. Nevertheless, despite its broad use, artificial aeration of lake water is still fairly controversial due to different effects on nutrient variation and its subsequent effects on total algal biomass. A 10-year long-term study showed that there were no effects of DO on nutrient concentration by artificial destratification (Gachter and Wehrli, 1998). In contrast, other long-term studies showed a reduction of TP in bottom water; however, these study results were skewed by interannual variations and partial destratification (Grochowska and Gawrońska 2004; Hanson and Austin, 2012). Moreover, a decline of N and P concentrations were due to periodical release from the bottom sediments (Grochowska and Gawrońska, 2004). In addition to long-term studies, the short-term studies, for instance, a three year study by Cowell et al., (1987), showed a decrease of TP and orthophosphate in the water column in the first year of aeration and gradual increase of TP and orthophosphate in the second year of artificial destratification. Conversely, David et al., (2009) showed that orthophosphate concentration was suppressed under oxic conditions, which was likely due to ferric iron binding of orthophosphates. This suggests that orthophosphates concentration to the overlying water column is control by internal phosphorous loading from sediments. However, a seven year mixing at Lake Nieuwe Meer in Amsterdam showed no changes of TP and TN concentrations in the water column by artificial mixing (Jungo et al., 2001). In contrast, some studies showed an increase of nutrient concentration by artificial

destratification (Brosnan and Cooke, 1987; Osgood and Stiegler, 1990). Osgood and Stiegler (1990) showed a two-to-three fold increase of TP, total kjeldahl nitrogen, chlorophyll *a*, and a decrease of Secchi disk transparency during four year study at hypereutrophic Crystal lake in Minnesota. All these findings imply that there are no solid conclusive studies to confirm that oxic bottom sediments increase or decrease nutrient release from sediments to overlying water column. Therefore, a comprehensive study that measures N and P throughout the lake will provide information that managers can use to restore or maintain eutrophic lakes.

In addition, no studies have focused on 24-hour sampling under aerated and non-aerated conditions in eutrophic lakes. Algal growth is highly responsive to diurnal variations; therefore, conducting a 24-hour water quality monitoring would provide a better understanding about diurnal variations of dissolved oxygen, water temperature, and pH at artificially aerated eutrophic lakes along with N and P concentration variations.

1.3. Hypotheses and Objectives

1.3.1. Hypotheses

1. Aeration will result in complete mixing and will eliminate thermal stratification of the lake, which will improve dissolved oxygen concentration in the near-bottom water.
2. Under aeration, dissolved inorganic P will be released by both aerobic decomposition of suspended organic materials and Fe-desorption. Further, dissolved inorganic nitrogen, as ammonia gas, will be released only by aerobic decomposition of suspended organic materials. Under aerated conditions, both dissolved inorganic N and P will be slowly and continuously released by microbial decomposition of suspended organic matter, but dissolved inorganic P released by Fe-desorption will be terminated under non-aerated conditions. So, under aerated conditions, the rate of N and P release will increase with

time compared to non-aerated conditions; therefore, more dissolved inorganic N and P will be expected in the water column under aeration. Therefore, it is hypothesized that aeration will increase nutrient concentration including nitrogen and phosphorous in the overlying water column.

1.3.2. Objectives

The main objective of this study is to investigate whether artificial aeration increases or decreases nutrient concentration in the water columns of lake and the impact of aeration on algal growth. This goal will be accomplished by:

1. Investigating the effects of aeration on water quality across the entire impoundment by performing water quality monitoring at multiple locations and across the water column under the conditions of aeration;
2. Determining nutrient concentration variation in the overlying water column under the condition of aeration and non-aeration; and,
3. Identifying the source (s) of nutrients, which are directly available for algal growth under the conditions of aeration.

CHAPTER 2. LITERATURE REVIEW

2.1. Eutrophic Lakes

2.1.1. Lake Trophic Status

Trophic status of a lake depicts biological productivity of the lake (EPA, 2010) and can be classified into oligotrophic, mesotrophic, and eutrophic. These categories reflect nutrient and clarity level of a lake. As shown in Figure 2.1, oligotrophic water bodies generally have low biological activity and excellent water quality; since the water contains low nutrient concentration and algae density. As the nutrient supply increases, the water body is described as mesotrophic and eutrophic. Mesotrophic water bodies lie in between oligotrophic and eutrophic stages and have medium biological activity and good water quality. However, when mesotrophic water bodies become eutrophic, the water body is rich with nutrients and excessive biological activity and poor water quality. The water is rich with nutrients and high rates of primary production of algae and aquatic plants and is susceptible to oxygen depletion in the near-bottom layer of lakes (Yang et al., 2008). Therefore, the conversion of a water body from oligotrophic to eutrophic causes severe environmental problems (excessive plant growth, fish kills, depletion of O_2 in the water column and low light penetration), which need to be addressed to restore a eutrophic lake.

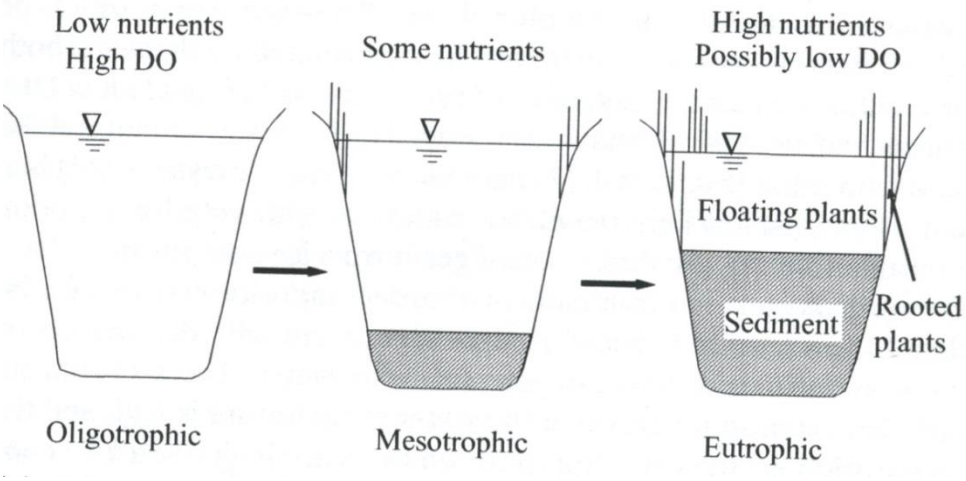

Figure 2.1: Eutrophication process from oligotrophic to eutrophic (Adapted from Ji, 2008)

Although nutrients are essential to algae, excessive nutrient levels, especially N and P, can be harmful to ecosystems because excessive concentrations cause eutrophication and stimulate the growth of aquatic weeds and algae. Bioavailable nutrients are mainly inorganic and are in dissolved form, including ammonium, nitrite, nitrate, and orthophosphate. Therefore, they are readily assimilated by plants including algae. Growth of phytoplankton is optimal at a C: N: P molar ratio of 106:16:1, which is referred to as the Redfield ratio (Redfield et al., 1963). The ratio of N: P is often used to assess the nutrient limitation for algal growth because algal growth is limited by either one or the other nutrient in the water body (Ryding and Rast, 1989). Total and bioavailable N: P ratios provide useful information regarding nutrient limitation; since the bioavailable N: P ratio changes through the growing season, the TN: TP ratio is used to determine overall conditions in the water body (Ptacnik et al., 2010). Indicator values of N: P ratios for phytoplankton growth are shown in Table 2.1. Vollenweider (1987) initiated the quantitative classification of trophic status based on N and P; however, the organization of Economic Cooperation and Development (OECD, 1982) expanded trophic status calculation by adding chl-*a* and transparency in the indicator value. Nitrogen, phosphorous, chl-*a* , and

transparency are now used as indicators for evaluation of lake eutrophication. The organization of Economic Cooperation and Development (1982) proposed the following threshold values for eutrophic lakes: TP concentration >0.035 mg/L; chl- *a* >0.008 mg/L; and transparency <3m.

Table 2.1: The optimum Nitrogen: Phosphorous ratios for phytoplankton growth. Dissolved inorganic nitrogen: Soluble reactive phosphorus (DIN: SRP) and Total nitrogen: Total phosphorus (TN: TP).

Source	Ratio	Nitrogen limited	Phosphorous limited	Reference
Water	DIN:SRP	<13	>50	Kosten et al., 2009
Sediment	TN:TP	<9	>23	Hecky et al., 1993

2.2. Lake Stratification

During warmer periods of the year as the surface waters are heated by solar radiation and become less dense and thermally resistant to wind mixing, lakes become stratified into three zones, such as the epilimnion, the hypolimnion and the metalimnion (Wetzel, 2001; Kalff, 2002). The epilimnion is the uppermost layer, which contacts the atmosphere. It is less dense and frequently turbulent water, and this is the zone where primary production is dominant. The lower water zone is more dense, cooler and largely nonturbulent and separated from contact with the atmosphere. The transition zone between the surface and deep water column is called the metalimnion and shows marked thermal change between the eplimnion and the hypolimnion therefore it is referred to as thermocline. At the end of the thermal stratification when loss of heat exceeds heat inputs, the eplimnion is mixed with the hypolimnion by wind and convection currents and eventually the entire lake volume mixes. So lakes are classified based on the stratification followed by periods of mixing or turnover.

2.2.1. Mixing Patterns of Lakes

Lakes can be categorized based on their mixing regime. According to the Hutchinson-Löffler classification system for temperate lakes, a period of summer and inverse winter stratification with an ice cover that are deep enough to stratify are categorized into amictic, dimictic, monomictic and polymictic (Lewis, 1983; Kalff, 2002). Amictic (not mixed) lakes, which are permanently covered by ice and lakes which have two mixing periods (spring and fall) once a year are called dimictic lake, while lakes where at lower latitude are designated as monomictic and mixes once annually. Polymictic (multiple mixing) lakes mix more than twice a year, ice covered during winter and those are relatively shallow or very wind exposed lakes (Wetzel, 2001; Grzybowski, 2014). Periods of mixing and thermal stability affect cycling and availability of dissolved inorganic nutrients to phytoplankton, light availability, water temperature variations and eventually primary production and lake's food web structure (Wetzel, 2001; Taranu et al., 2010). In eutrophic lakes, primary productivity is mainly a function of phytoplankton biomass and depends upon light availability; however, in polymictic lakes, light can severely limit primary productivity due to turbidity by algal biomass or suspended solids by wind-exposure (Gervais and Behrendt, 2003). Further, a combination of wind resuspension, periodic anoxia on calm days, and macrophyte decomposition add to the nutrient recycling in polymictic lakes. Moreover, in polymictic lakes, due to frequent wind exposure, nutrients that are released from the sediment are made available to euphotic zone rapidly in contrast to the lakes that have strong thermal stratification in summer, which slows or prevents nutrient transport from sediment to the overlying water column (Welch and Cooke, 1995).

2.3. Mechanisms of Lake Eutrophication

2.3.1. External Supplies of Nutrients

The external supplies of nutrient in a lake are derived from a wide variety of sources, including groundwater, atmospheric, and fluvial (discharges from rivers), and those nutrients can originate as point sources, which are localized and easily monitored and controlled (waste water effluent, runoff and leachate from waste disposal sites, runoff from construction sites and runoff and infiltration from animal feedlots), and as nonpoint sources, which are much more difficult to monitor and regulate (runoff from agriculture and runoff from pasture and rangelands) (Carpenter et al., 1998). Excess supply of agricultural fertilizers to land may accumulate in soil, move from the land into surface waters and migrate into ground waters or enter the atmosphere via ammonia volatilization (Carpenter et al., 1998). Nitrogen deposited from fossil fuel combustion in a watershed may contribute substantially to the nonpoint source pollution of surface waters Nutrient exports from both point and nonpoint sources can have profound effects on quality of receiving water and impairment of aquatic ecosystems by eutrophication and can have substantial economic effects (loss of biodiversity, increased recreation and angling cost, decreased lake property values and increased drinking water treatment cost) (Dodds et al., 2009).

2.3.2. Internal Loading of Nutrients

Internal loading of nutrient is the process in which organic particulate matter is decomposed to dissolve nutrients by oxidation and deoxidation. Lakes have different processes of internal nutrient release. In a deep, stratified lake, the mechanism for nutrients in the sediment bed to return to the water column is via diffusion in the form of dissolved nutrients. The lake is thermally stratified, so the thermocline acts as a barrier to nutrient upward transport; therefore, a lake turnover may mix the bottom nutrient throughout the water column. In contrast to deep

lakes, in shallow lakes the sediment bed and the water column play a major role in determining the nutrient concentration, water clarity and algal biomass. In shallow lakes, nutrient concentration in the water column may change due to the following interactions: resuspension of sediments by current and wind wave, sorption and desorption of nutrients from resuspended sediments, diffusion exchange of nutrient and uptake and release of nutrients by macroinvertebrates (bioturbation) on the lake bottom (Selig et al., 2003; Havens et al., 2003). Internal loading is the major contributor to P concentration in the water column as well as lake trophic status (Moore et al., 1998; Mardesen 1989). Fisher et al., (2005) studied the relative contribution of sediments to the nutrient budget of Lake Okeechobee, Florida using various sediment types. They found that sediment diffusive fluxes are a significant source of bioavailable N and P, specifically dissolved reactive phosphorous and dissolved inorganic nitrogen.

2.4. Factors Affecting N and P Release from Lake Sediments

There are several physical, chemical, and biological factors that influence the nutrients release from sediments to the overlying water column, such as water temperature, pH, dissolved oxygen (DO) concentration, chemical diffusion, bacterial activity and mineralization processes, bioturbation, iron to phosphorous ratio and contribution of macrophytes (Bostrum et al., 1982; Sondergaard et al., 2003; Christophoridis and Fytianos, 2006). Each of these contributing factors are discussed in the following sections.

2.4.1. pH

The influence of pH on sediment P release has been shown to play an important role in nutrient release from a lake sediment (Jensen and Anderson, 1992; Moore and Reddy, 1994; Eckert et al., 1997; Penn et al., 2000; Christophoridis and Fytianos, 2006; Jin et al, 2006). Moore and Reddy (1994) and Penn et al., (2000) found that soluble reactive phosphorous (SRP)

concentration was high (18 mg P L-1) under acidic (pH 5.5) conditions due to dissolving of hydroxyapatite and the subsequent release of adsorbed P. However, several laboratory and field studies showed that high P release rates from sediments can occur under high pH value (pH from 8 to 12) (Jensen and Anderson, 1992; Eckert et al., 1997; Christophoridis and Fytianos, 2006; Jin et al, 2006) due to the replacement of PO43- anions with OH- at high pH. Therefore, both high and low pH could influence sediment P release rates, while no P release has been shown to occur under neutral conditions (Jin et al., 2006).

2.4.2. Water Temperature

Nutrient release mechanisms are also linked to water temperature and biological activities (Jensen and Anderson, 1992). Increased water temperature stimulates release of inorganic phosphate via microbial degradation of organic matter (Gale et al., 1992; Jensen and Anderson 1992; Holdern and Armstrong, 1980). In addition, Jensen and Anderson (1992) demonstrated the impacts of water temperature on P release under aerobic conditions using sediment cores collected from four lakes (Arreskov, Sobygard, Kvind, and Vaeng) in Denmark. The water temperature range used for the study was 7˚C - 14˚C and 21˚C. This study showed the increase of SRP and O_2 uptake with the increase of temperature and the decrease of thickness of the oxidized surface layer. In addition, the study suggested that the increase of SRP was probably because of organic matter degradation.

2.4.3. Dissolved Oxygen

In addition to pH and water temperature, DO also affecting nutrients release from sediments. The oxygen concentration in sediments greatly affects biogeochemical processes. Decomposition of organic matter is affected by the DO concentration in the water column and biological oxygen demand (BOD) of the suspended sediment. Moore et al., (1992) observed that

organic matter decomposition is related to various O_2 levels (0, 0.2, 2 and 20) in sediment samples collected from the Lake Apopka, a hypereutrophic hardwater lake located in central Florida. This result showed that the highest amount of CO_2 released at 20% of O_2 treatment, indicating that decomposition was stimulated under high O_2 supply conditions.

2.4.4. Chemical Diffusion and Bioturbation

The interstitial water of the sediment is also important for the phosphorous transport between sediment and water (Bostrum et al., 1982). An upward transport of phosphorous is created via a diffusion-mediated concentration gradient normally appearing just below the sediment surface. Diffusion is generally considered the main transport mechanism from anaerobic hypolimnetic sediments where water movements are small above the sediment surface (Hesslein, 1980). The rate of a diffusional flux is primarily determined by the concentration gradient between interstitial water and lake water and the more pronounced the gradient, the faster the nutrient release rate (Kamp-Nielsen, 1974).

Bioturbation, is the mixing of sediment at the sediment-water interface by burrowing, feeding, resuspension, secretion, excretion and transporting activities of benthic organisms and/or benthivorous fish, which alter the structure and properties of the sediment thereby influencing transport of nutrients (N and P) from sediments to the overlying water column (Meysman et al., 2006; Kristensen et al., 1992). Sediments often have a lower oxygen concentration than the overlying water due to decomposition of organic matter by microorganisms. The activity of bioturbing animals increases oxygen penetration in the sediments, which enhances the mineralization process by which nutrients are released in to the overlying water column. The presence of invertebrates, such as chironomids and bristle worms, may directly contribute to the phosphate transport from sediment to water column via excretion

(Phillips et al., 1997). In addition, mixing of sediment during common carp (*Cyprinus carpio*) feeding activity resulted in increased nutrient levels, turbidity and seston particles (Phillips et al., 1997).

2.4.5. Resuspension

In shallow lakes, wind-induced resuspension is a mechanism that frequently causes increased concentrations of suspended solids in the lake water. Particulate bound forms of phosphorous settling to the bottom may be resuspended several times before permanent sedimentation (Ekholm et al., 1997). Sediment resuspension is considered to be a significant source of nutrients for phytoplankton growth (Carper and Bechman, 1984). During sediment resuspension, NH_4^+ released upon desorption, can be rapidly oxidized to NO_3^-. Different studies have shown aerobic P release under field conditions and laboratory conditions. For example, Holmroos et al., (2009) studied the effect of resuspension on aerobic release of phosphorous using experimental water columns in the shallow, eutrophic Kirkkojarvi basin of Lake Hiidenvesi in southern Finland. The authors found increased TP concentrations in all experiments, and 46% of the phosphorous was in the form of SRP and affected the phytoplankton growth by providing bioavailable phosphorous. In addition, many studies have shown the effects of resuspension on SRP release under laboratory conditions (Reddy et al., 1996; Sondergaard et al., 1992). In addition to release of P under low redox potential, studies have shown that aerobic release of P from sediments into the overlying water can be substantial (Hupfer and Lewandowski, 2008).

2.5. Mechanisms of N and P Release from Lake Sediments

2.5.1. Adsorption/Desorption

There are two main mechanisms for nutrient release from sediments to the overlying water column in lakes: Fe desorption and microbial decomposition (Gale et al., 1992; Christophoridis and Fytianos, 2006; Hupfer and Lewandowski, 2008).

Adsorption is the process where phosphorous bound to ion is removed from the solution, and desorption refers to the release of phosphate from the solid phase into solution phase. Phosphorous adsorption has been identified as an important mechanism for P removal from the aquatic ecosystem (Mortimer 1941; Froelich et al., 1982). Phosphorus adsorption could occur by formation of chemical bonds through ligand binding and chemisorptions (Reddy et al., 1996). The reactive components, which are responsible for adsorption of phosphates, are clay, oxides and hydroxides of ion and aluminum, calcium carbonate and organic matter (Yan et al., 2008).

Adsorption of P by iron compounds depends upon environmental conditions, such as aerobic and anaerobic conditions, while aerobic or anaerobic conditions do not affect the ability of aluminum compounds to sorb phosphorous. Under aerobic conditions, a surface oxidized sediment layer overlies a reduced, thick sediment layer. In the reduced sediment layer, Fe (III) is reduced to Fe (II) and provides phosphates to the overlying oxidized surface layer. Then Fe (II) oxidizes to Fe (III) and forms phosphorous bound Fe (oxy) hydroxides (FeOOH~P), which are insoluble compounds. Under anaerobic conditions, phosphates are released from phosphorous bound Fe (oxy) hydroxides to the overlying water column. Therefore, in oxic conditions, P is chemically bound to insoluble Fe (III) compounds, while in reduced conditions Fe (III) is reduced to soluble Fe (II) compounds, and subsequently, both iron and sorbed phosphates return to solution (Bostrum et al., 1982).

The pH is important in lake sediments P-release because the phosphorous binding capacity decreases with increasing pH as hydroxyl ions replace orthophosphate ions (Jin et al., 2006). Christophoridis and Fytianos (2006) showed the effect of redox potential and pH on the P release from sediment using two lakes. This study was carried out under three redox potentials, -200mV, +100mV and +300mV, while the pH values used for the study were 7, 8 and 9.

Both lakes showed high P release rates under a reductive condition (-200mV) and high pH value (pH 9). The study showed that the increase of P release under a reductive condition was due to dissolution of P-bearing metal oxides (Fe oxides and hydroxides) and the replacement of PO_4^{3-} anions with OH^- at high pH. Moore and Reddy (1994) found an increase of P release under acidic conditions because of the dissolution of hydroxyapatite, while Christophoridis and Fytianos (2006) observed P release under high pH condition due to dissolution of Fe (PO_4).

Under anaerobic conditions, sulfate reduction occurs, part of the Fe^{2+} precipitates as FeS and dissolved phosphates diffuse from the sediment to the overlying water column because of dissolution of $Fe_3(PO_4)_3.8H_2O$ (Vivianite), which provides Fe^{2+} and phosphates to the water column (Eq. 1) (Caraco et al., 1993).

$$Fe_3(PO_4)_2.8H_2O + 3H_2S \longrightarrow 3FeS + 2H_3PO_4 + 8H_2O \qquad \text{(Eq.1)}$$

2.5.2. Microbial Decomposition

In addition to Fe desorption, another method for nutrients release is due to microbial decomposition of organic matter. Decomposition of organic matter is an essential process in the lake ecosystem, and through the decomposition process, nutrients in organic materials are converted to bioavailable forms, which are directly taken up by plants. Several factors influence the decomposition rates, such as O_2 supply, sediment type and temperature.

Microbial decomposition occurs under both aerobic and anaerobic conditions (Depinto and Verhoff, 1977; Gale et al., 1992; Moore et al., 1992). In aerobic decomposition, microbes use O_2 as the ultimate electron acceptor, while in anaerobic decomposition, NO_3^-, SO_4^{2-}, HCO_3^-, MnO_2, and FeOOH can be used as electron acceptors (DeLaune and Reddy, 2005). Organic matter decomposition under anaerobic conditions can be considered a stepwise process. In the first step, cellulose is converted to simple sugars. In the second and third steps, organic acids are formed and finally, organic acids converted to CO_2 and CH_4 (Neue and Sharpenseel, 1984). Conversely, end products of aerobic decomposition are CO_2 and H_2O, while under anaerobic decomposition the end products are CO_2 and CH_4 (Gale et al., 1992).

Ammonia release to the water column under anaerobic conditions is the result of ammonia build up in sediments due to two microbiological mechanisms: 1) loss of biological nitrification because nitrifying bacteria are aerobic and those bacteria are responsible for the oxidization of NH_4 into NO_2 and NO_3 (Rysgaard et al., 1994) and 2) a decrease in ammonia assimilation by anaerobic microorganisms (Beutel, 2006). Since anaerobic bacteria have much lower growth rates and cell yields than heterotrophic aerobic microorganisms, this results in the accumulation of ammonia in sediments (Moore et al., 1992).

Under oxic conditions, ammonia is released by organic matter decomposition and oxidized aerobically by the nitrifying bacteria such as *Nitrosomonas* and *Nitrobacter*. *Nitrosomonas* sp. oxidize ammonia to nitrite and another group, *Nitrobacter* sp., oxidize nitrite to nitrate (Madigan, et al., 1997). Two enzymes are involved in the oxidation of ammonia to nitrite, and the oxidation occurs in two steps: 1) bacteria use ammonia monooxygenase to convert NH_3 to NH_2OH and H_2O (Eq. 2) and 2) hydroxylamine oxidoreductase is used to oxidize

NH_2OH to NO_2^- (Eq.3). In nitrite-oxidation, bacteria use the enzyme nitrite oxidase to oxidize nitrite to nitrate (Eq.4) (Madigan, et al., 1997).

Ammonia- Oxidizing

$$NH_3 + O_2 + 2e^- + 2H^+ \longrightarrow NH_2OH + H_2O \qquad \text{(Eq.2)}$$

$$NH_2OH + H_2O + 1/2\ O_2 \longrightarrow NO_2^- + 2H_2O + H^+ \qquad \text{(Eq.3)}$$

Nitrite-Oxidizing

$$NO_2^- + 1/2\ O_2 \longrightarrow NO_3^- \qquad \text{(Eq.4)}$$

In addition to O_2 supply levels, sediment type also influences microbial decomposition of organic matter in the sediment. Gale et al. (1992) found that the rate of biodegradability of organic matter in relation to sediment type under anaerobic conditions was as follows: surface unconsolidated (loosely aggregated sediments) sediment materials > underlying consolidated (thick sediments) sediment materials > peat. Surface unconsolidated sediment materials, comprised of recently, deposited detrital organic matter are the readily decomposable fraction, while peat is the most resistant to decomposition.

In addition to O_2 supply levels and sediment type, temperature also influences the rate of carbon mineralization. Gudasz et al. (2010) found mineralization of organic carbon in lake sediments exhibits a positive ($r^2 = 0.61$) relationship with temperature using eight lakes in central Sweden. Further, authors suggested that warmer water temperature lead to more mineralization of organic carbon.

2.6. Restoration of Eutrophic lakes

2.6.1. Consequences of Lake Eutrophication

Lake eutrophication is the enrichment of two essential nutrients, N and P; therefore, as shown in Figure 2.2 an increase of these elements inevitably causes the increase of phytoplankton biomass and occurrence of algal blooms. These dense algal blooms and dead biomass severely deplete DO in the near- bottom layers and form an anoxic zone in the near-bottom. Anoxia as a result of eutrophication has major impacts on commercial and recreational fisheries.

Aquatic organisms need oxygen to survive and anoxic or hypoxic conditions negatively affect their life (Ji, 2008). The amount of DO is a sign of possible pollution because as DO level in water drops, aquatic life is put under stress. To maintain healthy aquatic life, the daily minimum DO requirement is 5 mg/L in the water column (NDDoH, 2012); however, the amount of DO to prevent mortality differ among species and for the various life stages; for instance, fish eggs and young fish are more sensitive than adults. Further, the oxygen concentration drops to less than 2 mg/L for a long period cause major fish kills coupled with diurnal DO changes mainly during highly productive eutrophic lakes in summer (NDDoH, 2012). Apart from the direct effects, there are also other important consequences of amount of oxygen in the water column, such as DO concentration affects nutrient releasing mechanisms; for instance, Fe desorption and microbial decomposition, which directly affects to stimulate algal growth and eventually lead to lake eutrophication. Further, decomposition of organic matter under anaerobic conditions produces undesirable odors, tastes and colors that impair the use of water body for their designated purposes (for recreation or drinking); for instance, fermentation process releases CH_4 and H_2S from sediment bed to overlying water column (Ji, 2008). In addition, ammonia is

reported to be toxic to freshwater organisms and presence of ammonia also depends upon environmental conditions; for example, under oxic conditions, ammonia is released and oxidized aerobically by bacteria and provide dissolved inorganic nutrients into the water column; however, under anaerobic conditions ammonia builds up in sediments due to denitrification and it may serve as the main mechanism for removing nitrate from hypolimnetic water.

In addition, some phytoplankton species produce toxins, and these blooms are called harmful algal blooms (HABs), which cause destruction of economically important fisheries (Burkholder et al., 1992), public health risks (Morris, 1999), and degradation of water quality. Furthermore, algal blooms limit light penetration, diurnal DO changes and reduction of aesthetic value. Among phytoplankton groups, some cyanobacterial species, such as *Microcystis* sp, *Anabaena* sp, *Oscillatoria*, and *Aphanizomenon* sp. (Ferreira et al., 2001; Mohamed and Shehri, 2010), are associated with HABs and tend to dominate nutrient rich fresh water ecosystems due to their superior competitive abilities under nutrient rich conditions (Paerl and Paul, 2012).

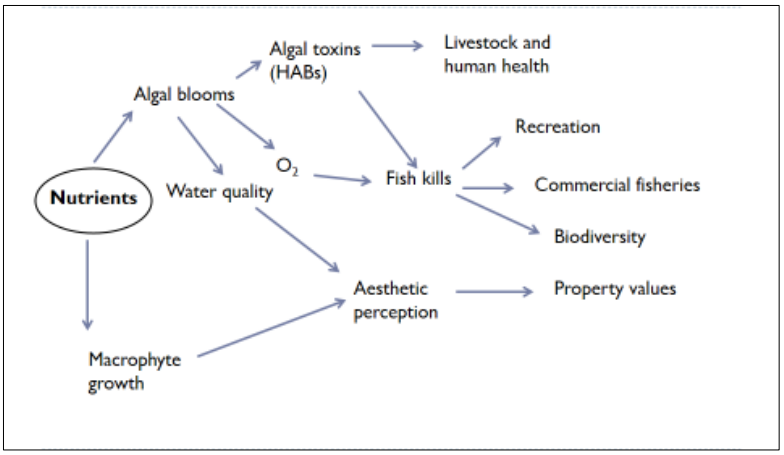

Figure 2.2: Impact of nutrient enrichment in a lake (Adapted from Harper, 1992)

2.7. Management Strategies for Lake Eutrophication

Several lake restoration techniques have been established, such as external nutrient loading control (e.g. Waste water treatment facilities, regulatory policies to reduce run off),

internal source control (sediment dredging, water flushing, nutrient precipitation, and aeration techniques) (Annadotter et al., 1999; Madgwick, 1999; Sondergaard et al., 2000) or ecosystem recovery (macrophyte implantation and biomanupulation) (Carpenter et al., 1995; Ha et al., 2013). Restoring eutrophic water bodies to their original conditions using ecological techniques takes a long time; however, ecosystem recovery technologies are low investment technologies.

The key measurement for the control of eutrophication is reducing the external nutrient loads, including point and non-point source pollution; however, nutrients can gradually accumulate in the bottom sediments and can subsequently be released to the overlying water to supply essential nutrients for algal growth. Therefore, Lauridsen et al. (2003) have shown that when external nutrient sources are controlled, still lakes can maintain their trophic status due to nutrient (P) release from lake sediments. The study also showed that reduction of external nutrient loading alone is insufficient for the restoration of water quality due to internal nutrient loading from bottom sediments (Lauridsen et al., 2003) Therefore, controlling internal loading of nutrients is essential in effective lake restoration.

The control methods have included a variety of lake restoration methods, such as increasing nutrients export (nutrients can be reduced by using the bottom outlet to drain away the nutrient-rich hypolimnetic water during thermal stratified conditions of the lake) (Klapper, 2003), increasing the P-sedimentation (phosphorous precipitation with iron salts, such as $FeCl_2$, $FeCl_3$ and $FeSO_4$ or alum salts, such as $AlCl_3$ and $Al_2(SO_4)_3$ to lower the lake's P content by removal of P from the water column) (Klapper, 2003; Galvez-Cloutier et al., 2012), aeration by physical mixing and bio manipulation (aquatic communities are manipulated to maintain phytoplankton biomass at low levels by removing planktivorous fish or by stocking piscivorous fish to reduce the planktivores fish) (Klapper, 2003; Sondergaard et al., 2007).

2.7.1. Artificial Aeration

Among lake restoration methods, aeration is typically used to increase dissolved oxygen in the near-bottom waters. The techniques employed to solve hypolimnetic anoxia can be grouped into hypolimnetic aeration, hypolimnetic oxygenation and artificial destratification (Beutel, 2006).

2.7.1.1. Hypolimnetic Aeration

Hypolimnetic aeration is a technique that is used to maintain oxic hypolimnia while preserving thermal stratification. As shown in Figure 2.3, the technique uses a confined air-lift system where air bubbles are injected at the bottom of an air-lift tube, and oxygen is transferred to the water as the air-water mixture travels up the tube. Hypolimnetic aeration may not work properly if the water body is too shallow; therefore, hypolimnetic aeration is not recommended if maximum depth of the water body is less than 12 to 15m (Cooke et al., 2005).

Studies have shown that lakes have been subjected to hypolimnetic aeration to improve dissolved oxygen in the near bottom layers. For example, Soltero et al., (1994) used partial and full lift hypolimnetic aeration systems at Medical Lake in Washington State, USA, to study the operation of two different aeration systems on physical (water temperature), chemical (dissolved oxygen, nitrate-nitrogen, nitrite-nitrogen, ammonia-nitrogen, soluble reactive phosphorous, and total phosphorus) and biological (chlorophyll *a* and phytoplankton biovolume) lake water quality parameters under varying oxygen concentrations. The authors found that operation of both aeration systems decreased the hypolimnetic ammonia and total phosphorous concentrations, which were in the near bottom water column. Similarly, Ashley (1983) also used a full lift hypolimnetic aeration system at Black Lake, in the Southern Interior Plateau limnological region

of British Columbia and found decreased hypolimnetic ammonia and hypolimnetic orthophosphate concentrations in the overlying water column.

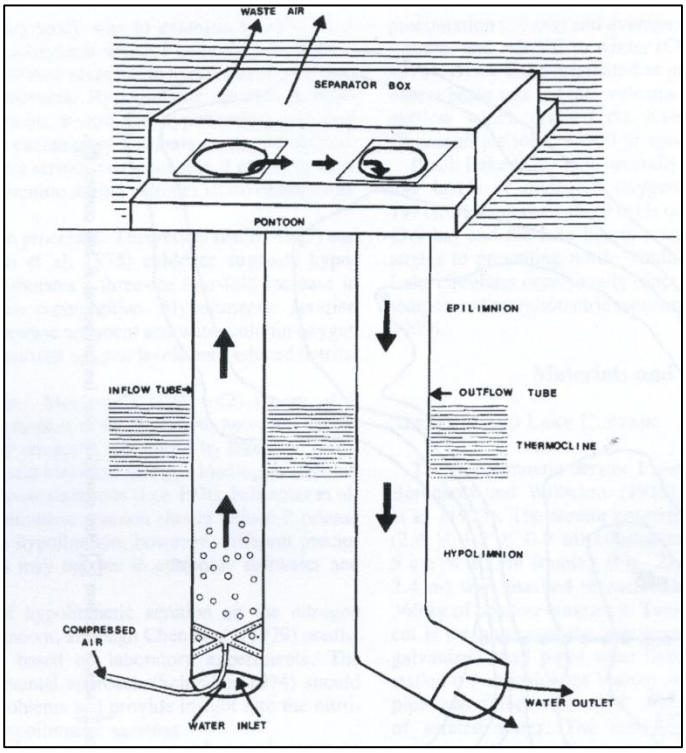

Figure 2.3: Schematic diagram of hypolimnetic aerator (Adapted from Ashely, 1983)

Advantages of hypolimnetic aeration include maintenance of stronger thermal stratification and increased operational flexibility. Another benefit of hypolimnetic aeration is that DO could be increased in the hypolimnion while maintaining normal cold water environment. Disadvantage of the system can be the cost. According to Pastorak (1982), to aerate Tegeler See Lake (which is 420 ha), by hypolimnetic aeration with fifteen aerators to deliver 4.5 tons of O_2 per day, the total initial cost of the aeration system was $ 2,732, 000 (1990 dollars). Another disadvantage is mixing in the hypolimnion may increase diffusion of nutrients, include N and P, upward through the thermocline even though stratification is maintained (Beutel, 1999, 2002 & 2006).

2.7.1.2. Hypolimnetic Oxygenation

Hypolimnetic oxygenation is the newest method used to increase the DO in the near-bottom layers. Like hypolimnetic aeration, hypolimnetic oxygenation also prevents the formation of thermal stratification; however, hypolimnetic aeration uses air, while hypolimnetic oxygenation uses pure oxygen (Beutel, 2002). There are three types of systems currently in use: side bubble plumes, line diffusers and submerged contact chambers (Figure 2.4). For example, Prepas and Burke (1997) have studied the effects of hypolimnetic oxygenation on water quality in Amisk Lake, Alberta, at the deepest area of four study basins and they found decreased summer hypolimnetic total phosphorous concentration and hypolimnetic ammonium concentration.

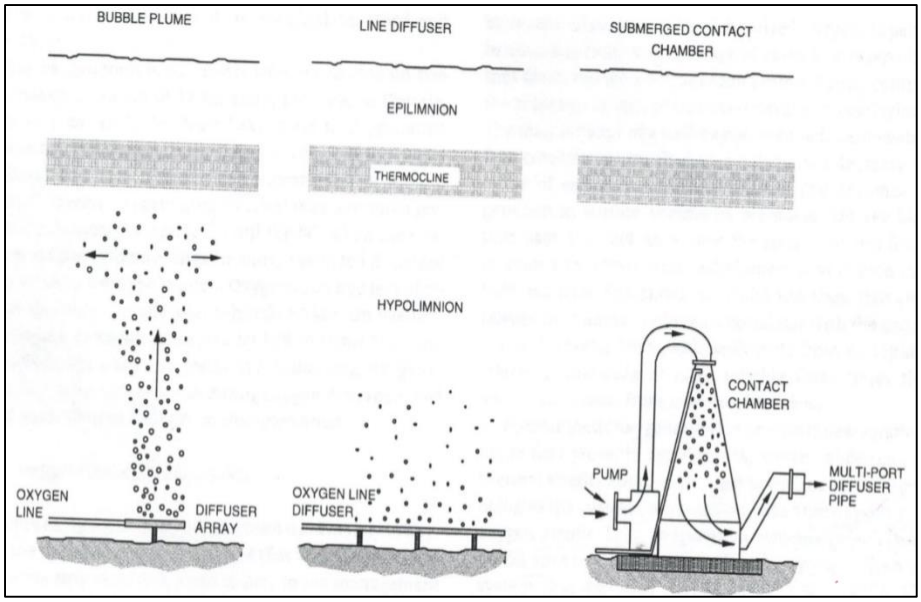

Figure 2.4: Schematic of three common hypolimnetic oxygenation systems (Adapted from Beutel, 2002)

In addition, a 10-year study of combined artificial mixing and hypolimnetic oxygenation in two eutrophic Swiss lakes showed that internal P cycling was not affected by increased hypolimnetic dissolved oxygen concentration (Gachter and Wehrli, 1998). An overall reduction

in P concentration was observed and found to be caused by temporal changes in external loading during the study. Additionally, the authors concluded that the reduction of P may also be due to the precipitation of phosphates under anoxic conditions prior to the start of the aeration.

The main advantages of hypolimnetic oxygenation is higher oxygen solubility; therefore, the size of mechanical devises and recirculation rates needed to deliver an equivalent amount of oxygen using pure oxygen rather than air are greatly reduced. Due to lower recirculation rates, minimal turbulence is introduced into the hypolimnion (Beutel, 2006). Disadvantage of the hypolimnetic oxygenation system can be the cost. For example, operating cost for various oxygenation systems per day may range from $850 to $3,000, liquid oxygen may cost $150/ metric ton, and energy costs of $0.10/ kWh (Beutel, 2002).

2.7.1.3. Artificial Destratification (Artificial Circulation)

In artificial destratification, compressed air is injected through pipes or diffusers located at the bottom of the water column. Rising air bubbles produce vertical mixing, which prevents the formation of thermal stratification (Beutel, 2006). Injection of compressed air at maximum depth usually affords the greatest rate of mixing (Cooke et al., 2005). Lorenzen and Fast (1977) concluded that an air-flow rate per lake surface area of 9.2 m^3/min/ km^2 should provide complete destratification of a lake. Optimal destratification performance cannot be expected at lower air flow rates (Hanson and Austin, 2012). Destratification increases DO in the near-bottom layers through redistributing oxygen from surface to bottom, and increasing the contact time between water surface and the atmosphere. Compared to hypolimnetic aeration and oxygenation, the benefit of re-oxygenation is to increase the suitable habitat for aerobic animals in lakes.

Several studies illustrate the use of artificial destratification system in lakes. Heo and Kim (2004) collected water samples from Lake Dalbang (a reservoir located in the north eastern

region of South Korea) and they found that destratification did not affect the regular seasonal pattern of P concentration in the overlying water. However, Cowell et al. (1987) carried out a study at Lake Brooker in northwest Hillsborough County, Florida, and found that annual mean TP and SRP concentrations decreased from June, 1981 to May, 1982 under destratified condition and again increased from June, 1982 to June, 1983 while aeration was in progress.

There are several factors that need to be considered when designing an artificial aeration system. For example, Ashley (1987) cited an annual cost of $202, 000 (in 2002 US dollars) for two air compressors producing an air flow rate of 9.2 $m^3/min/km^2$, which was needed for complete destratification of thermally stratified lakes. The cost included pipes and air diffusers. Another consideration is that, the depth at which compressed air is released is critical to solve the low dissolved oxygen problem in the near-bottom layers. To avoid this problem, placing air diffusers at the lake bottom is recommended (Pastorak, 1982). For the complete destratification of lakes, the flow rate of air per unit of lake surface should be about 9.2 $m^3/$ min/km^2 and within the range of 6.1 to 12.3 $m^3/$ $min/$ km^2 (Ashley,1987). As shown in Figure 2.5, in the process of destratification by a rising plume of air bubbles, water is pulled from the hypolimnion into the epilimnion. The thermocline breaks up, which results in homothermous, completely mixed conditions near the plume.

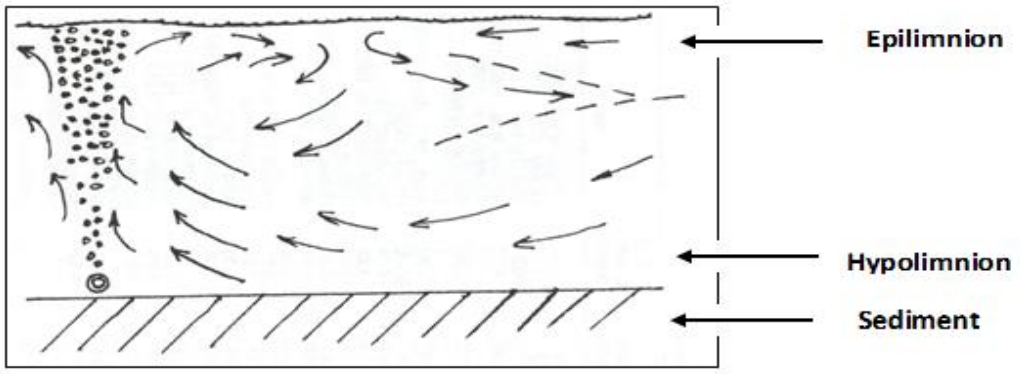

Figure 2.5: Vertical circulation of air by a rising plume of air bubbles (Adapted from Ji, 2008)

CHAPTER 3. MATERIAL AND METHODS

3.1. Description of the Study Area

The Heinrich Martin Dam Impoundment (HMDI) is located in north central LaMoure County, ND, and it contains a small impoundment and earthen dam (Figure 3.1). The HMDI is a small reservoir with a surface area of 18.8 acres (0.08 km^2), and it is used for recreational activities such as boating and fishing (Wax et al., 2008). The average and maximum depths of water are 4.3 m and 10 m, respectively (Wax et al., 2008). The main water supply for the impoundment is runoff from surrounding agricultural lands, which are dominated by small grains and row crops, and maximum water level is controlled by an overflow structure (Overmoe, 2008).

Figure 3.1: Areal view of HMD

3.2. The HMD Aeration System

The aeration system of HMD is an artificial destratification system, which includes a 45.72m long and 12.7 mm diameter aeration tube connected with the supply line placed in the deepest area (10 m) of the reservoir (Overmoe, 2008). The HMD artificial aeration system (Air diffusion system-A John Hinde Co.) was installed in May, 2006 by the North Dakota Game and Fish Department (NDG&F), and it was powered by a 220 volt Sweetwater ¾ horsepower oil-less piston pump for the purpose of breaking down the thermal stratification, increasing the DO concentration in the impoundment, and improving fish growth (Overmoe, 2008). The pressure of the pump ranged between 18 and 28 PSI from 2006 to 2008 (Kratz, 2007). The aeration system was in operation since 2006 (with the exception of 2009) in early summer prior to lake stratification for 5-6 months per year.

The cost associated with the aeration system operation from 2006 to 2008 were: in 2006, the total cost for maintenance and electricity was $385 for 148 days: in 2007, it was $380 for 164 days, while in 2008, the cost was $436 for 145 days (Kratz, 2007).

Figure 3.1: The HMD aeration system (Photograph from NDDoH)

3.3. Sampling Schedule

Water samples were obtained with nonmetallic Van Dorn vertical water sampler and sampler was lowered slowly to the sampling depth. Samples were collected from June 4th to

October 15th in 2010, June 30th to November 8th in 2011, and June 20th to October 3rd in 2012. Six sampling sites (A, B, C, D, G, and H) were selected in the impoundment based on effects from aeration system and depth variability (Table 1; Figure 3.2). Site A is located in the deepest part of the impoundment and is near the air diffuser; Site B is also near the air diffuser but further from site A; sites C and D are away from the air diffuser and located in the western and eastern arm of the impoundment, Site G is closer to the inlet, and Site H is located in the dead area. Table 1 contains the GPS coordinates and depth of each sampling location.

Table 3.1: Sampling locations at the HMD

Sampling point	**A**	**B**	**C**	**D**	**G**	**H**
Longitude (W)	98.5362	98.5360	98.5379	98.5347	98.5344	98.5385
Latitude (N)	46.5923	46.5929	46.5930	46.5937	46.5954	46.5930
Water depth:m	8	6.5	4	5	1.5	1.5

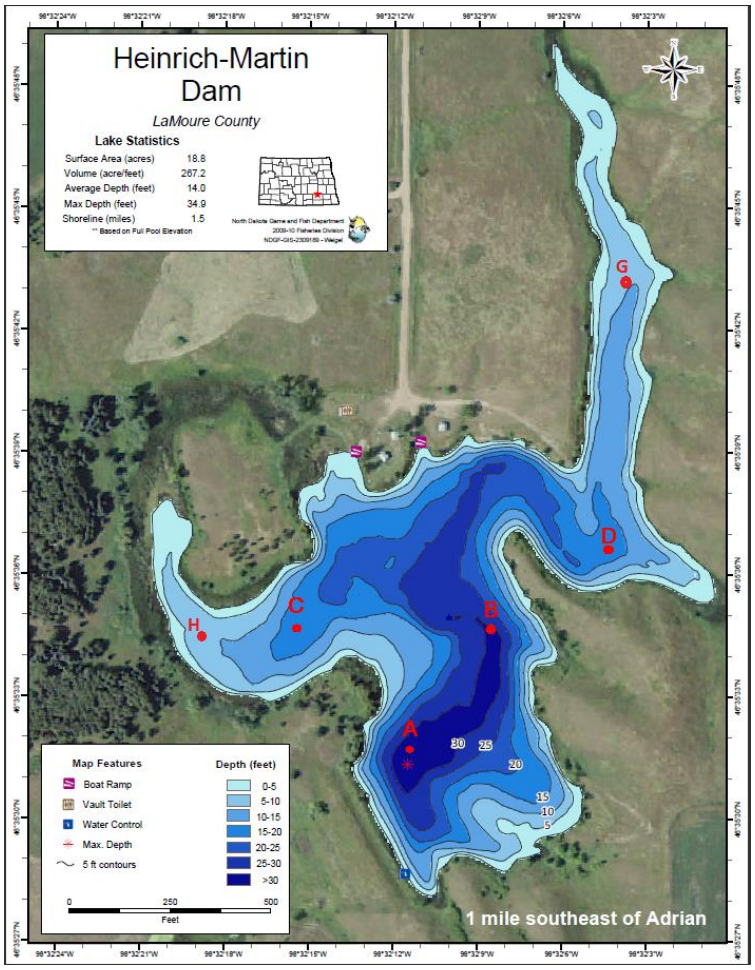

Figure 3.2: Sampling location sites A, B, C, D, G and H at HMD

Water quality monitoring (nitrate-nitrogen, nitrite-nitrogen, ammonia-nitrogen, total nitrogen, total dissolved nitrogen, soluble reactive phosphorous, acid hydrolysable phosphorous, total phosphorous, chlorophyll a, dissolved oxygen, water temperature, turbidity, Secchi depth, total suspended solids and volatile suspended solids, conductivity and pH) and water sampling were carried out on a weekly and biweekly basis in 2010, 2011, and 2012 for better understanding of algal population variation and nutrient concentrations in the impoundment. In 2010, water quality monitoring and sampling were carried out under the conditions of aeration at sites A, B, C, and D on a biweekly basis. In 2011, samples were collected on a biweekly basis during an aerated period and weekly during the period without aeration (from July 13th to

September 1st) at sites A, B, C, and D. In 2012, in addition to four sampling locations (A-D), two new locations, sites G and H, were added to study the impacts of aeration on DO and nutrient concentration in the water column. Moreover, sampling was carried out under the conditions of aeration during the sampling period except a few days when the aeration system did not work properly.

To determine the vertical nutrient variations across the depths, water samples were collected from four different depths using Van Dorn vertical water sampler. The sampler was lowered slowly to the depth to be sampled, which were Secchi depth, 2× Secchi depth, 1.5 m from the bottom, and 0.5 m from the bottom under the conditions of aeration. Those depths were selected to determine the presence of phytoplankton and to determine the vertical gradient of nutrients in the water column. Based on transparency of light through the water column, Secchi depth and 2× Secchi depth were varied during the study period. Water samples were taken at five depths (0.5m from the surface, Secchi depth, 2× Secchi depth, 1.5m from the bottom, and 0.5 m from the bottom) for chl *a*, total suspended solids (TSS), and volatile suspended solids (VSS) analyses at each site under the conditions of aeration. In addition, water samples were taken at Secchi depth, 2× Secchi depth, 1.5 m from the bottom, and 0.5m from the bottom and analyzed for nutrients under the conditions of aeration.

In 2010, samples for nutrient analyses were taken only from Site B since the impoundment was aerated, and we assumed that the nutrient distribution was homogenous. In 2011, when the impoundment was aerated, only one sample for nutrient analyses from Secchi depth was taken at each site. In 2012, water samples were collected for nutrient analyses at sites A-D at the following depths: at 0.5 m from the surface, at Secchi depth, at 2× Secchi depth, 1.5m from the bottom, and at 0.5 m from the bottom under the conditions of aeration. Samples for

nutrient analyses were only collected at 0.5 m from the surface at sites G and H because these sites are very shallow areas. In addition, samples were taken only at surface and Secchi depth for chl-*a* because most of the phytoplankton were reported in the epilimnion. Therefore, samples were collected at 0.5 m from the surface and Secchi depth at sites A-D and only at surface at sites G and H. Water samples were taken using a nonmetallic Van Dorn vertical water sampler, which was lowered slowly to the depth to be sampled.

In addition, after aeration system had been turned off, additional samples were taken at Secchi depth, 1 m below the bottom of the thermocline, 1.5 m from the bottom, and 0.5 m from the bottom at deep location sites A and B and at Secchi depth at shallow location sites C and D for N and P species analyses under non-aerated conditions. Further, samples were also taken at 0.5 m from the surface, Secchi depth, 1m below the bottom of the thermocline, 1.5 m from the bottom, and 0.5 m from the bottom at deep locations sites A and B, and at 0.5 m from the surface, Secchi depth, and 0.5 m from the bottom at shallow location sites C and D for chl-*a* analyses and turbidity measurements under non-aerated conditions.

Further, weather data were downloaded from the nearest weather station located in Marion, LaMoure, ND, USA, (NDAWN), about 16 km (10 miles) northeast of the HMD.

3.3.1. On-site Measurements

Dissolved oxygen, water temperature, pH, Secchi depth, turbidity, and electrical conductivity (EC) were measured in the field. Dissolved oxygen, water temperature, pH, and EC were monitored at near-surface waters (0.5 m below the water column) and 1m depth intervals throughout the water column at each site using a Yellow Spring Instruments (YSI) multisensor sonde to determine vertical variations across the depths. The measurement error for temperature was estimated to be ± 0.15°C; for conductivity was estimated to be 0.5% of measured value; for

DO was estimated to be ± 2% of reading (ranging from 0 to 20 mg/L dissolved oxygen); for pH was estimated to be ± 0.2 units; and for turbidity was estimated to be ± 2% of measured value. Dissolved oxygen, pH, and conductivity sensors were calibrated in the laboratory before each field sampling, following the standard operating procedure recommended by the manufacturer. Turbidity was measured using a 2100P Turbidimeter at 0.5 m from the surface, Secchi depth, 2× Secchi depth, 1.5m from the bottom, and 0.5 m from the bottom at all four sites in 2010, 2011, and 2012, and turbidity was measured only at surface in sites G and H. Secchi depth was measured using a Secchi disk at all four locations in 2010, 2011, and 2012.

3.4. Total Suspended Solids and Volatile Suspended Solids

The total suspended solids (TSS) were measured using a well-mixed sample, which was filtered through a GF/C (pore size is 1.2 µm, 47 mm diameter) glass fiber filter and the residue retained on the filter was dried to a constant weight at 103-105ºC. The standard deviation was estimated to be 5.2 mg/L (coefficient of variation 33%) at 15 mg/L, 24 mg/L (10%) at 242 mg/L, and 13 mg/L (0.76%) at 1707 mg/L total suspended solids (APHA, 2005).

The volatile suspended solids (VSS) were measured after ignite the residue produced by TSS in a muffle furnace at a temperature of 550ºC. The VSS measurements were taken after a constant weight was obtained. The standard deviation was estimated to be 11 mg/L at 170 mg/L volatile total solids (APHA, 2005).

3.5. Chlorophyll *a* Analysis

Prior to chlorophyll *a* analysis, water samples were filtered through GF/F (pore size between 0.5 to 0.7 µm, 47mm diameter) glass-fiber filters to separate algae from water samples. Filter papers were transferred into 20 ml bottles and 5 ml of 95% ethanol was added, and those bottles were incubated in the dark for 20h to extract chlorophyll pigments into organic solvent

(APHA, 2005). The extracted solution was filtered with GF/F glass-fiber filter to reduce the turbidity and remove any filter paper particles. Then absorbance was measured spectrophotometrically at 750 nm and 665 nm (Lorenzen, 1967; Sartory et al., 1984). All extractions were carried out in the dark to minimize degradation of chlorophyll pigments because chlorophyll could further degrade into chlorophyllides, pheophorbides, and pheophytins (APHA, 2005). The measurement error for chlorophyll was estimated to be ± 0.55 µg/L.

3.6. Analytical Methods

Water samples were analyzed for inorganic nitrogen species, including ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-); total nitrogen (TN); soluble reactive phosphorous (SRP); total phosphorous (TP); and total acid hydrolysable phosphorous (AHP). Nutrient determinations were made according to the American Public Health Association (APHA, 2005). Samples were filtered through 0.45 µm membrane filters except for TN, TP and AHP analyses. Soluble reactive phosphorous concentration was determined spectrophotometrically, according to the 4500-P/ascorbic acid method (APHA, 2005). Photometric accuracy was estimated to be 1% at 0.50-2.0 absorbance for spectrophotometric analyses. Ammonium was measured using 4500-NH_3 /phenate method and NO_3^--N was analyzed using the 4500-NO_3^-/nitrate electrode method (APHA, 2005). The measurement error was estimated to be ± 0.4 mv, corresponding to 2.5 % in nitrate-nitrogen concentration for nitrate electrode method (APHA, 2005). Nitrite was measured by the UV spetrophotometric method (APHA, 2005). Total phosphorous was determined after digestion with sulfuric and nitric acid followed by spectrophotometric analysis as for SRP. Total nitrogen was determined after 4500-N/persulfate digestion followed by nitrate electrode method as for NO_3^--N. Acid hydrolysable phosphorous was determined after acid hydrolysis followed by spectrophotometric analysis as for SRP. The relative standard deviation was estimated to be 9.1

% at 100 µg/L, 4.0 % at 600 µg/L, and 5.2 % at 7000 µg/L orthophosphates for ascorbic acid method (APHA, 2005).

3.7. Twenty-four hour Water Quality Monitoring

In 2011, a twenty-four hour water quality monitoring was carried out at non-aerated Site A to study the influence of algal growth on diurnal DO variation in water column. Dissolved oxygen, DO saturation, water temperature, pH, and conductivity were measured at one meter depth intervals in every three hour interval using an YSI 650 multi-parameter sonde (YSI Incorporated, Yellow Springs, OH, USA).

In 2012, a twenty-four hour water quality monitoring was carried out under the conditions of aeration to compare the influence of algal growth on diurnal DO concentration with non-aerated conditions. In addition to DO, water temperature, pH, and conductivity were also measured every 15 minute interval using automated system of YSI 650 multi-parameter sonde.

3.8. Statistical Analyses

One way analysis of variance (ANOVA), least square mean analysis, and Tukey mean comparison were used to evaluate for differences in water quality parameters across the sampling depths, across the sampling dates, and across the sampling locations. Furthermore, ANOVA was used to examine changes of means for the water quality parameters between non-aerated period (July and August) and aerated period. When a significant difference ($p \leq 0.05$) was found, a multiple comparison technique (least significant difference) was used to determine between which year(s) it occurred. All statistical test were performed using SAS 9.3 version and statistical significance was considered to be $p \leq 0.05$.

CHAPTER 4. RESULTS AND DISCUSSION

4.1. Dissolved Oxygen (DO)

Dissolved oxygen balance, nutrient concentration variation, and algal growth are correlated, and understanding the relationship between these parameters provides vital information to lake managers needing to address lake eutrophication. Figure 4.1 shows summer DO isopleths for the deepest location, site A, in 2010, 2011 and 2012 over depth and time. During these three summers, only at the surface were DO concentration above 5 mg/L (minimum DO standard for warm water lakes under NDDoH water quality standards). However, DO was above 2 mg/L in near bottom layers under aerated conditions in 2010 (Figure 4.1a). During the non-aerated period in 2011, DO did exceed the standard value at the surface (Figure 4.1 b) but was between < 2 mg/L in near bottom at the deepest location on most sampling dates that year (Figure 4.1b). In contrast, during 2012 DO exceeded was above the standard value at the surface on some of the sampling dates, but dropped to ≤1 mg/L in near the bottom on two dates (July 6th and August 27th) when the aeration system did not work properly (Figure 4.1c).

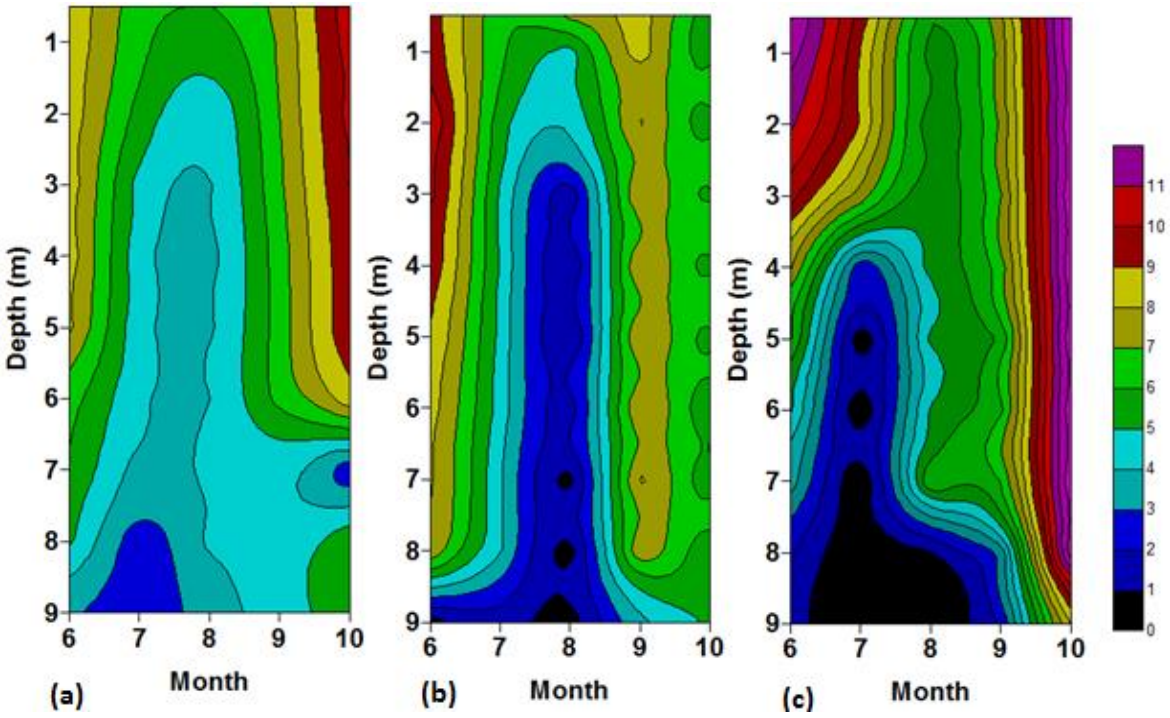

Figure 4.1: The DO (mg/L) isopleths for the deepest location (site A) (a) in 2010 under aerated conditions, (b) in 2011 under non-aerated conditions and (c) in 2012 under aerated condition. In 2011, the non-aerated period was only between months 7 and 8. In 2012, the aeration system did not work properly during early July (July 6th), late August (August 27th) and early September (September 4th)

Figure 4.2 shows DO variation under aerated conditions (2010) versus non-aerated conditions (2011) at the deepest location. The non-aerated period in 2011 spanned only July and August (months 7 and 8). In Figure 4.2, a DO gradient is shown by contour lines that are widely spaced reflecting vertical mixing caused by aeration. In contrast, during lake stratification the DO gradient under non-aerated conditions is steeper (indicated by contour lines are more closely spaced), indicating high DO variability across the water column.

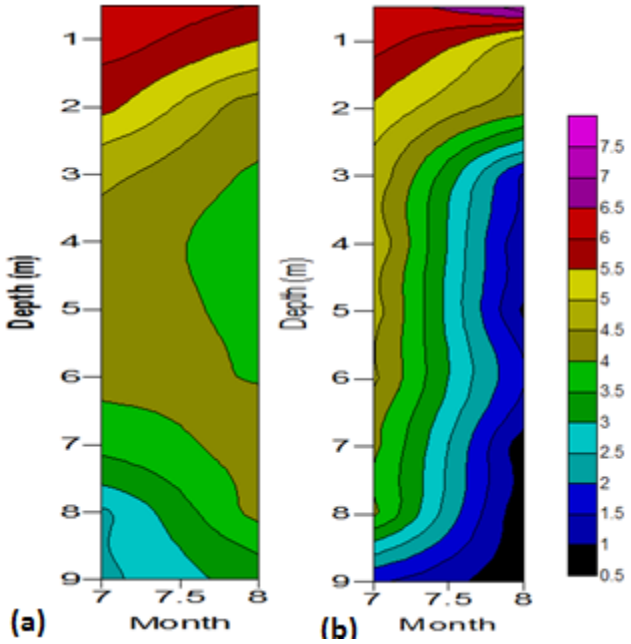

Figure 4.2: The DO (mg/L) isopleths for the deepest location (site A), (a) in 2010 under aerated conditions versus (b) in 2011 under non-aerated conditions

As shown in Figure 4.3, site C is shallow and wind mixing increases mixing depth and provides a subtantial amount of DO into the water column in addition to the effects of aeration; therefore, in 2010, DO was above 4 mg/L in near bottom layers at the shallowest location (Figure 4.3a) compared to near bottom layers at deep locations (Figures 4.1 and 4.2). In 2011, when the aeration system was turned off, DO dropped to less than 2 mg/L in near bottom layers at the shallowest location (Figure 4.3b), which was a similar concentration to deep locations. In 2012, since the aeration system did not work properly, DO did not increase as expected, and DO dropped to less than the recomended value even at the surface while aeration was in operation (Figure 4.3c).

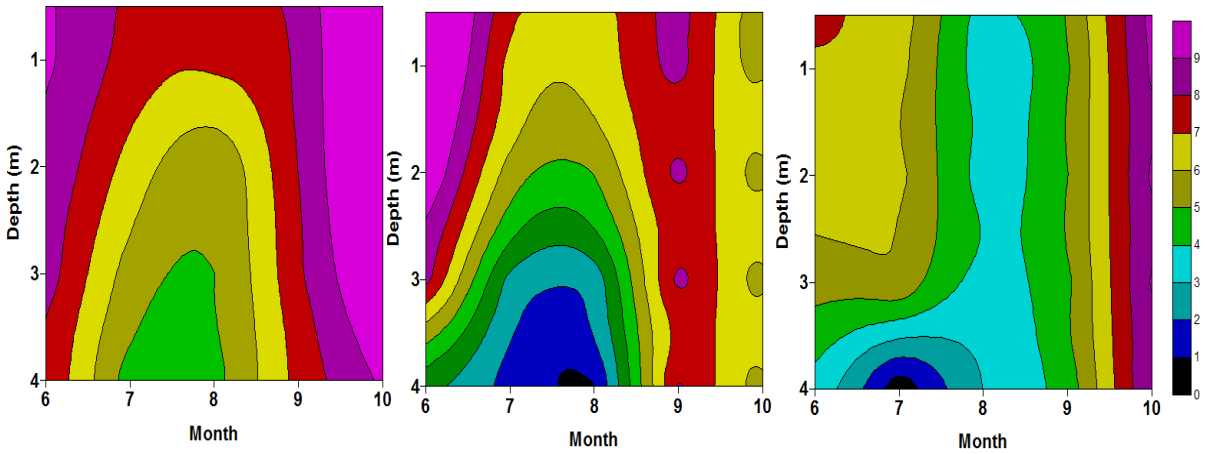

Figure 4.3: The DO (mg/L) isopleths for the shallowest location (site C), (a) in 2010 under aerated conditions, (b) in 2011 under non-aerated conditions and (c) in 2012 under aerated condition. In 2011, the non-aerated period was only between months 7 and 8. In 2012, the aeration system did not work properly during early July (July 6th), late August (August 27th) and early September (September 4th)

Moreover, Figure 4.4, which is similar to Figure 4.2, shows DO variation under aerated versus non-aerated conditions at the shallowest location. In comparison, under aerated conditions contour lines were more widely spaced and indicates DO differences across the water column was lower than the non-aerated conditions which contour lines showed a greater difference across water column when the impoundment was stratified after aeration was turned off.

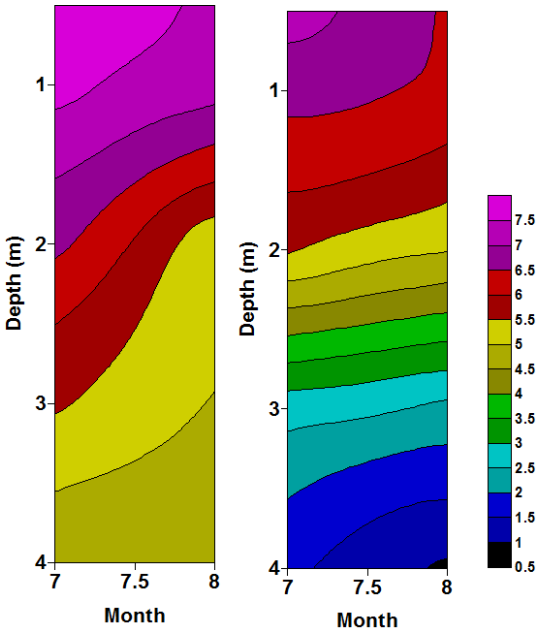

Figure 4.4: The DO (mg/L) isopleths for the shallowest location (site C), (a) in 2010 under aerated conditions versus (b) in 2011 under non-aerated conditions

The air diffuser was located near the deepeset location; therefore, DO variation at deepest location versus the shallowest location under aerated and non-aerated conditions provide crucial information about DO changes in response to depth variability. As shown in Figure 4.5, the DO was less than the daily minimum (5 mg/L) in most of the water column compared to the shallowest location which DO was above 5 mg/L in the whole water column.

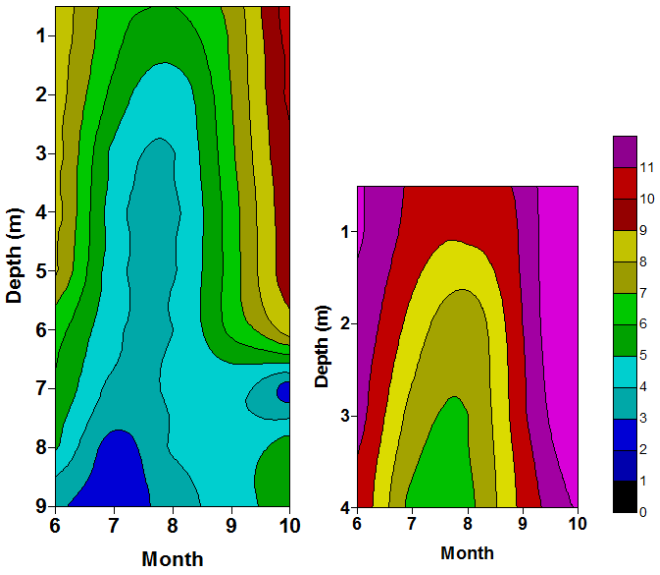

Figure 4.5: The DO (mg/L) isopleths for, (a) the deepest location versus (b) the shallowest location under aerated conditions in 2010

Moreover, similar to Figure 4.5, Figure 4.6 shows DO variation at the deepeset location versus the shallowest location under non-aerated conditions. Contour plots show more closely spaced contour lines regarless of the depth variability which indicates a greater variability of DO throughout the water column. During the non-aerated period, the top most layer separated from the undisturbed hypolimnion; therefore, DO production occurred only in the surface layer due to algal photosynthesis, and high DO consumption occurred in bottom layers. Among deep and shallow sampling locations under the non-aerated period, a prolonged period of low DO was observed at site A, which is the deepeset location of the HMDI (Figure 4.6a).

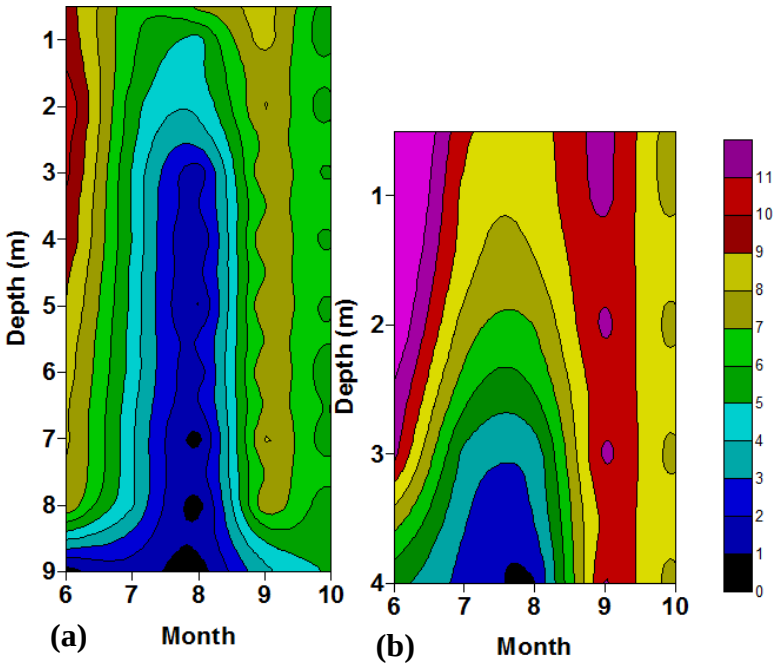

Figure 4.6: The DO (mg/L) isopleths for, (a) the deepest location versus (b) the shallowest location under non-aerated condition in 2011. The non-aerated period for both locations was only between 7 and 8 months

The DO solubility depends upon water temperature; therefore, DO variation can be explained partially by water temperature changes. Moreover, as shown in Figure 4.7, the DO concentration decreased gradually from June to July due to increasing water temperature; however, at the end of August when water temperature was decreasing, which increases solubility of DO. The DO saturation was calculated using the water temperature relavent to each sampling depth at each of the four locations during the sampling period. The DO saturation also followed the same trend of DO, and the water column was undersaturated throughout the sampling period until the end of October. The DO concentration at the surface of the water column was closer to the saturation line compared to the near bottom layers at all sampling locations.

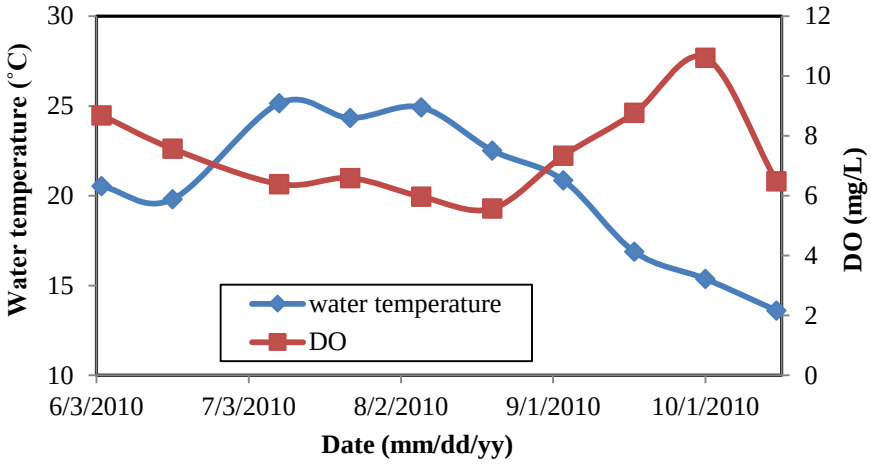

Figure 4.7: The surface DO concentration versus water temperature at the deepest location (site A) under aerated conditions in 2010

A previous study (Overmoe, 2008) showed higher biochemical oxygen demand (BOD) at the bottom layer of a deep location, which indicated a high concentration of organic matter in near bottom layers. In addition, the study showed that percent volatile organic matter content at the bottom sediment at the HMDI was between 14 and 19%, which was higher than organic matter content in fertile agricultural soils, which is between 3 and 6%.

Since the HMDI had high organic matter at the bottom layers when aeration was turned off, DO concentration rapidly dropped to less than 1 mg/L in near bottom layers during the non-aerated period (Figures 4.1 & 4.2). Furthermore, results indicated that due to high organic matter content in the near bottom layers, artificial aeration temporarily increases DO concentration in the water column, and aeration possibly redistributes sediment organic matter throughout the water column by resuspension. In 2010, when aeration was in operation, DO increased in the near bottom layers; however, in 2012, when the aeration system did not work properly, DO concentration rapidly dropped in the near bottom layers. Nevertheless, artificial aeration

eliminated thermal stratification in both 2010 and 2012 compared to the non-aerated conditions in 2011.

The comparison of DO concentration under pre-aerated conditions (2001-2005) and aerated conditions (2007-2010) in summer at the deepest location reveals that the aeration improved DO concentration within the impoundment (Figure 4.8); however, even with aeration, DO concentration gradient across depths was observed during the study period. The DO concentration during pre-aerated period was less than 5 mg/L, which is the daily minimum DO concentration except at 1 m depth; however, when aeration was in operation, DO was above daily average except in the near bottom layers in summer which indicated the aeration system has greatly enhanced DO concentration throughout the water column and expanded aerobic habitats for aquatic organisms. (Figure 4.8).

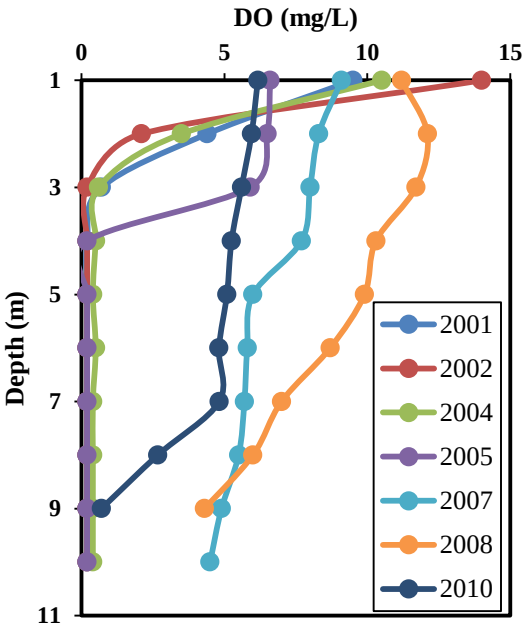

Figure 4.8: Dissolved oxygen variation during pre-aerated (2001-2005) versus aerated (2007-2010) conditions at the deepest location, site A (Data source: NDG&F)

4.2. Water Temperature

Water temperature is another important factor that influences lake eutrophication due to several reasons, including (1) DO solubility is largely determined by water temperature; (2) most biochemical processes, such as dissolved nutrients released by decomposition of organic matter by microorganisms, are governed by temperature (increased temperatures can increase decomposition rates) and; (3) many aquatic organisms can tolerate only a limited range of temperatures.

Figure 4.9 shows water temperature isotherms for the deepest location under aerated condition in 2010, under non-aerated condition in 2011, and under aerated condition in 2012. In 2010, under aerated condition, the water column showed an isothermal condition (Figure 4.9a), and no significant differences ($p<0.05$) in temperature observed throughout the water column; likewise, as shown in Figure 4.9b, when aeration was turned off in 2011, a weak thermal stratification was established in the impoundment, and no significant differences ($p<0.05$) were observed throughout the sampling depths. In 2012, when aeration was turned on, water temperature variation was more similar to 2010 and there were no significant differences ($p<0.05$) observed across the depths, which indicated that the water column was isotherm; however, since the aeration system did not work properly as expected, the water temperature at the bottom layer was 2-3 °C lower compared to the surface as non-aerated period in 2011(Figure 4.9c).

Moreover, the contour plots show a gradual increase of water temperature from early June to mid-July, which denoted with a change of color from green to pink, and a gradual decrease of water temperature from mid-July to October, which denoted with a change of color

from pink to black and contour lines are more closely arranged (Figure 4.9). Those closely spaced contour lines imply the presence of a temporal gradient of water temperature (Figure 4.9).

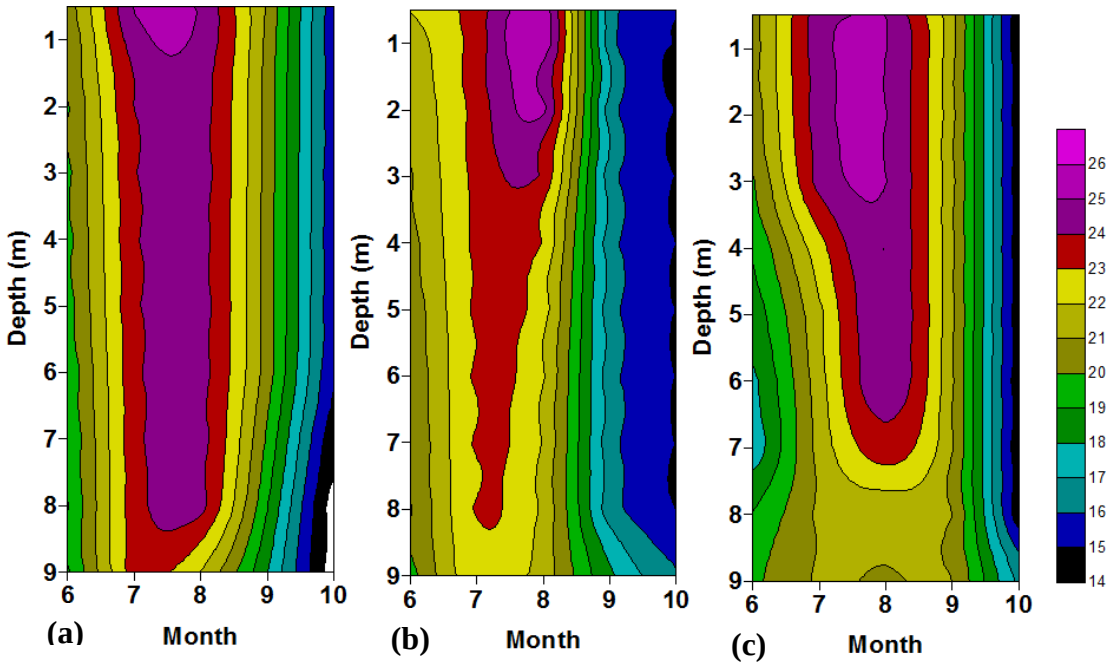

Figure 4.9: The water temperature (°C) isotherms for the deepest location (site A), (a) under aerated condition in 2010, (b) under non-aerated condition in 2011 and (c) under aerated condition in 2012. In 2011, the non-aerated period was only between months 7 and 8. In 2012, aeration system did not work properly during early July (July 6th), late August (August 27th) and early September (September 4th)

In addition, Figure 4.10 compares the water temperature variations under non-aerated condition, which was only 7 and 8 months in 2011, with aerated condition during the same months in 2010. It clearly shows under aerated condition, the water temperature was between 23 and 25°C and the impoundment was well mixed by the existing aeration system (Figure 4.10a); however, under non-aerated condition, the water temperature varied between 21 and 26°C which indicates maintenance of relatively cool hypolimnion during non-aerated condition (Figure 4.10 b).

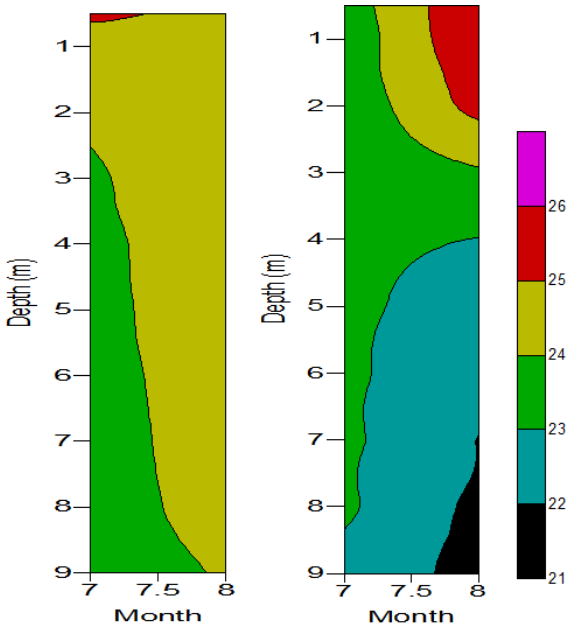

Figure 4.10: The water temperature (°C) isotherms for the deepest location site A, (a) under aerated condition in 2010 versus (b) under non-aerated condition in 2011

Similar to Figure 4.9, Figure 4.11 also shows the water temperature isotherms for the shallowest location. Under non-aerated condition, Figure 4.11b shows a weak stratification compared to aerated conditions in both 2010 and 2012 as shown in Figure 4.9.

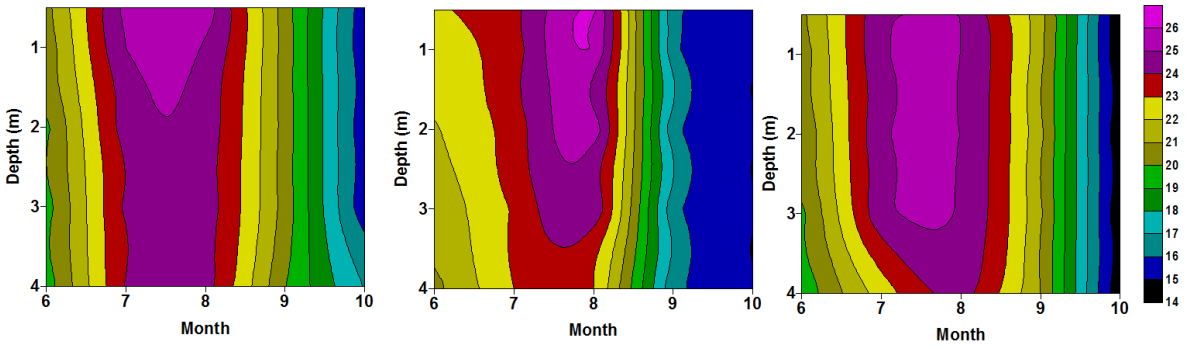

Figure 4.11: The water temperature (°C) isotherms for the shallowest location site C, (a) under aerated condition in 2010, (b) under non-aerated condition in 2011 and (c) under aerated condition in 2012. In 2011, the non-aerated period was only between months 7 and 8. In 2012, aeration system did not work properly during early July (July 6th), late August (August 27th) and early September (September 4th)

Moreover, Figure 4.12, which is similar to Figure 4.10, it is evident that partial mixing of water column during non-aerated condition (Figure 4.12a) compare to Figure 4.12b. In addition, Figures 4.13 and 4.14 show water temperature variation among the deepest location and the shallowest location under aerated and non-aerated conditions, respectively. The contour plots show no differences in water temperature in respect to depth under aerated conditions, and water column was mixed in both locations regardless of the depth. Statistical analyses also showed no significant differences ($p<0.05$) in water temperature across sampling locations.

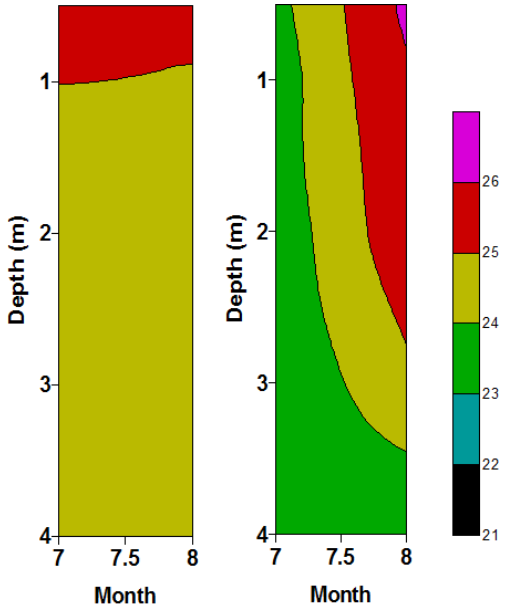

Figure 4.12: The water temperature (°C) isotherms for, (a) the shallowest location (site C) under aerated condition in 2010 versus (b) under non-aerated condition in 2011

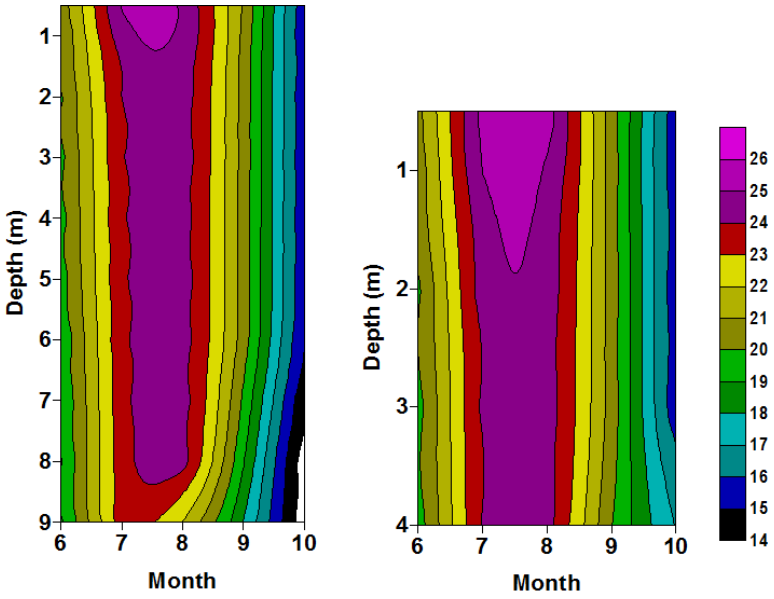

Figure 4.13: The water temperature (°C) isotherms for, (a) the deepest location versus (b) the shallowest location under aerated condition in 2010

Furthermore, as shown in Figure 4.14, water temperature variation at the deepest versus shallowest under non-aerated condition also showed no differences in water temperature for those two locations in respect to their depth. The difference of the water temperature in the bottom layers at those locations between 7 and 8 months was just 1°C. Further, in 2010, the mean water temperature throughout the water column was 24ºC at all sampling locations (sites A, B, C and D), while in 2011, the mean water temperature at the surface layer was 25ºC at all sampling locations; but due to a weak stratification under non-aerated conditions in 2011, in near bottom layers water temperature was 2-3 ºC less than the surface water temperature at all the sampling locations. In addition, contour plots for both the shallowest and the deepest locations show there was a temporal temperature gradient throughout the study period; however, there was no temperature gradient throughout the water column under aerated conditions, while under non-aerated conditions a weak thermal gradient occurred. A maximum limit for water temperature for

lakes and reservoirs in North Dakota is 29.44 ºC. The HMDI water temperature was less than this limit in all three years at all the sampling locations.

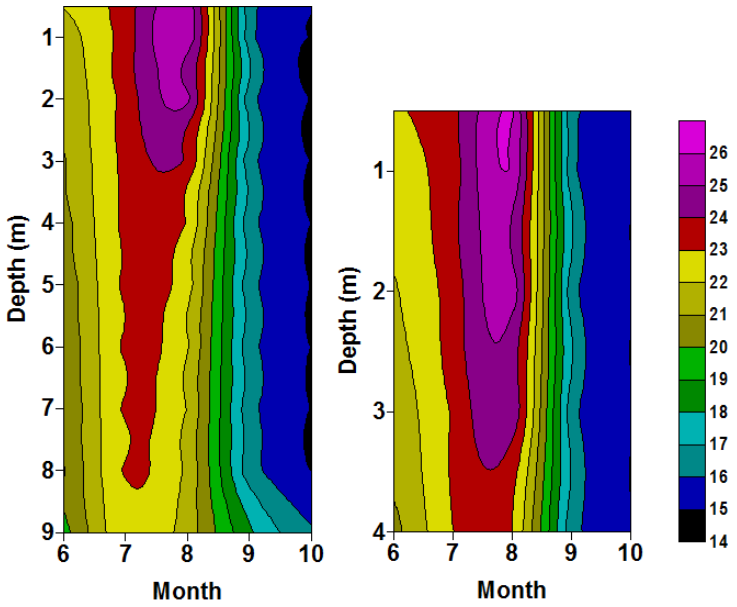

Figure 4.14: The water temperature (°C) isotherms for, (a) the deepest location versus (b) the shallowest location under non-aerated conditions in 2011. The non-aerated period for both locations was only between months 7 and 8

In addition to seasonal variations on water temperature changes, the daily water temperature could be changed due to wind effect. Wind mixing is susceptible to be an important factor in mixing of epilimnion in thermally stratified lake in summer. The lake warms as convection, summer storms mixes epilimnion more frequently and since the HMDI is a relatively shallow lake it may experience frequent mixing with winds, and such lakes are called polymictic lakes. According to the topography of the North Central region of the United States, a large fraction of the land area is well exposed to the wind, and the state of North Dakota, where the HMDI is located indicates a significantly greater wind energy potential (Elliott et al. 1986). Therefore, lakes in the state of North Dakota are exposed to more winds throughout the year and water quality, such as water temperature, possibly have influence from wind mixing in addition

to other controlling factors. Therefore, most likely in epilimnion could be enhanced by wind effects in addition to seasonal changes at HMDI.

In addition to nutrients, DO, and water temperature, the pH is also very important when considering lake management efforts since the HMDI supports a warm water fishery. The mean pH of the HMDI under both aerated and non-aerated conditions in 2010, 2011 and 2012 at all sampling locations was between 7 and 9, which was within the range of NDDoH water quality standards for lakes and reservoirs.

4.3. Variations of Conductivity under Aerated versus Non-aerated Conditions

Conductivity is a vital thermophysical property of water, and conductivity depends on dissolved ions concentration in the water. Information can be used as an indicator of water chemistry (index of total dissolved solids) (Moore et al., 2008) and indicator of hydrologic process. For instance, temporal variability of conductivity values can be used as an indicator of changing runoff sources (Moore et al., 2008). The major positively charged ions determine the conductivity are sodium, calcium, potassium and magnesium, while negatively charged ions are chloride, sulfate, carbonate, and bicarbonate; in addition, nitrates and phosphates are minor contributors to conductivity (Das et al., 2005).

In 2010, results showed that no significant differences ($p<0.05$) in conductivity throughout the water column and across sampling locations. Therefore, results imply that the water column mixed by aeration and dissolved substances distributed homogenously throughout the lake. Figure 4.15 shows average conductivity throughout the study period at HMDI under aerated conditions in 2010. In 2011, under the non-aerated period, the surface layer conductivity was unchanged and remained around 800 µS/cm; however, conductivity increased to 975 µS/cm in the bottom layer at the deepest location, site A (Figure 4.16). This increase in conductivity in

the near-bottom water indicated internal loading of dissolved substances during the period without aeration. No significant differences ($p<0.05$), however, were observed across depths in conductivity at sites A, B, C and D (Figure 4.17).

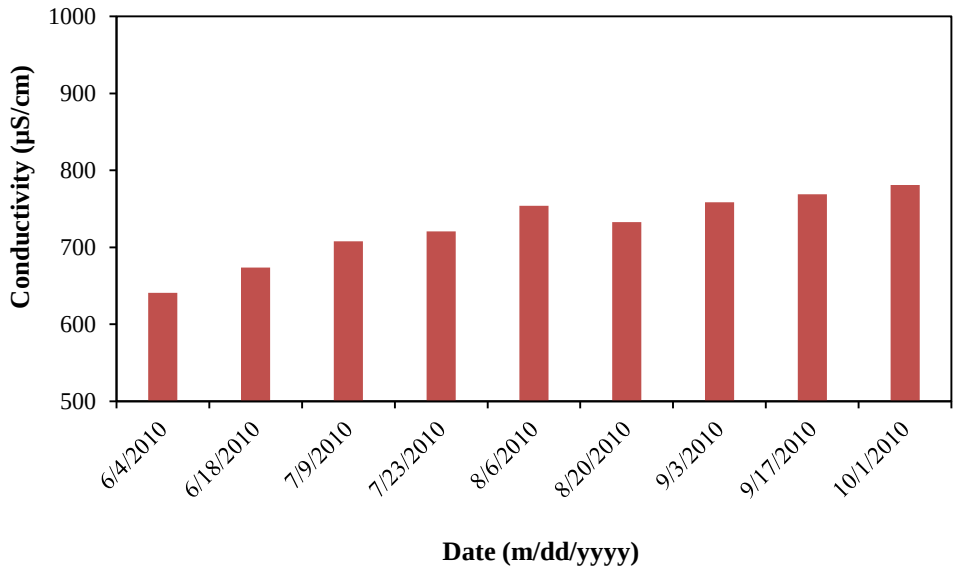

Figure 4.15: Temporal variations in conductivity at sites A, B, C and D under aerated condition in 2010

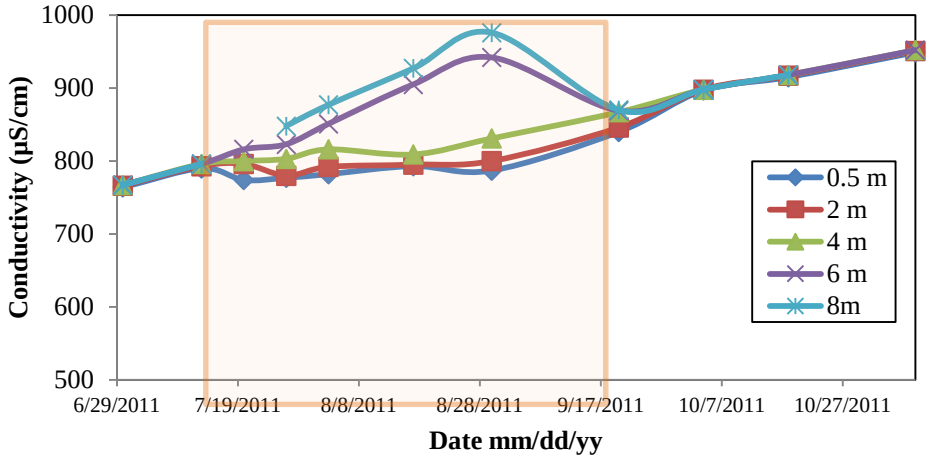

Figure 4.16: Conductivity temporal and vertical distributions at the deepest location, site A under non-aerated condition in 2011. The period without aeration is indicated by a square

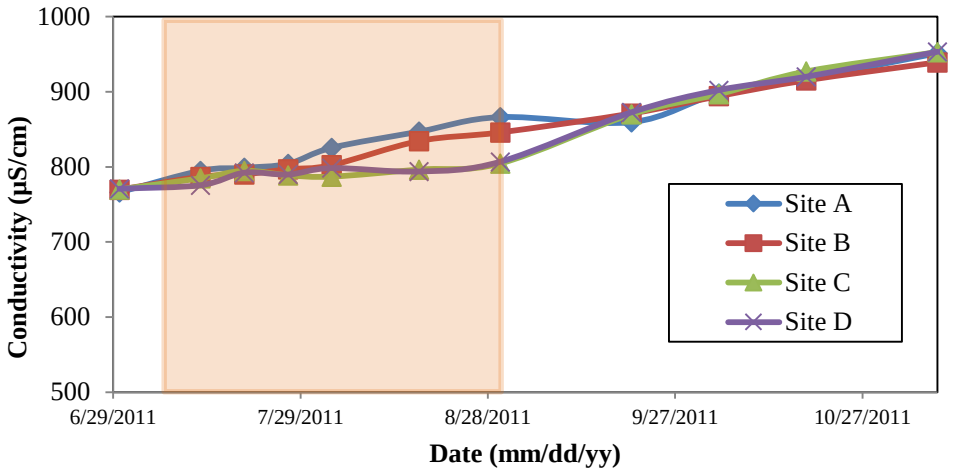

Figure 4.17: Temporal variation of conductivity at sites A, B, C and D under non-aerated condition in 2011. The period without aeration is indicated by a square

Similar to 2010, in 2012 conductivity showed a gradual increase with time and no significant differences ($p<0.05$) across sampling depths in any sampling location. Figure 4.18 shows temporal variation of conductivity at all six locations in 2012. Conductivity increased gradually from mid-June and slightly dropped in mid-August; however, again conductivity increased gradually by the end of sampling in mid-October at all locations. Results indicated that continuous supply of dissolved ions, including N and P, into the water column when aeration was in operation, and results showed no significant differences ($p<0.05$) across sampling locations, which imply dissolved ions are homogenously distributed across locations.

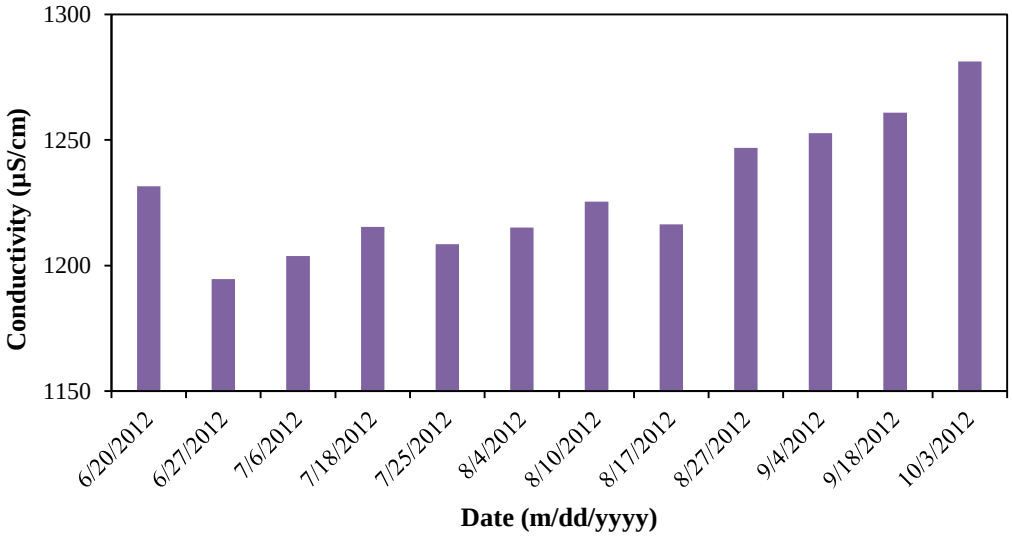

Figure 4.18: Temporal variation of conductivity at six sampling locations (sites A, B, C, D, G and H) under aerated conditions in 2012

The mean conductivity between July and August of the HMDI was approximately 730 µS/cm at both surface and Secchi depth at all locations (sites A, B, C and D) in 2010, while conductivity was slightly increased in 2011 and was 785 µS/cm at the surface in all locations in 2010; however, in 2012, conductivity was approximately 1215 µS/cm under the conditions of aeration. Conductivity at both surface and Secchi depth was significantly ($p<0.05$) high from 2010 to 2011, from 2011 to 2012 as well as from 2010 to 2012 at all sampling sites, implying that increased dissolved substances including dissolved inorganic nitrogen, phosphorous and dissolved organic substances derived from microbial degradation and other salts. Furthermore, results indicated conductivity was increased significantly over the study period; however, mean conductivity values at each sampling location were still less than the regional average of other reservoirs and impoundments in North Dakota.

The conductivity for lakes and rivers varied between 10 and 1000 µS/cm and average conductivity of reservoirs and impoundments in North Dakota was 1618 µS/cm; therefore, the

conductivity of the HMDI was in the range when comparing historical water quality data and below the level, which is 1000 µS/cm in 2010 and 2011 regardless of the aeration. Conductivity depends upon the presence of charged ions, water temperature and run off from surrounding agricultural lands. Temporal variability of conductivity observed during all three years indicated that increase in ions other than nitrates and phosphates or changing runoff sources most possibly dissolved ion concentrations in the water column due to seasonal lake overturn, and results suggested that contribution of dissolved nutrients into the water column by internal release of nutrients from the lake sediments. In addition, conductivity increased from 2010 to 2012 implies that possibly increase of dissolved ions concentration in the water column. There was not enough information available about changes of other ions such as carbonate and bicarbonate concentrations in the water column at the HMDI throughout the study period. In addition, the conductivity values did not fluctuate during the study period and no significant differences among open locations (sites A and B), shallow locations (sites C and D), the inlet (site D) and dead end area (site G) indicated there was no significant external contribution of dissolved ions into the water column.

4.4. Influence of Aeration on Algal Biomass

Primary producers, which include phytoplankton and macrophytes, utilize nutrients for their growth and play a key role in in the eutrophication process. Algae affect the nitrogen cycle, phosphorous cycle and the DO balance primarily through nutrient uptake, oxygen generation via photosynthesis and oxygen depletion following algal death. Since algae remove dissolved nutrients from the water column, nutrient uptake associated with algal growth is a crucial aspect of nutrient cycling in lakes. Algae take up dissolved inorganic nutrients and recycle these nutrients in the forms of organic phosphorous (OP) and organic nitrogen (ON). Plants use

sunlight is an energy source for photosynthesis, producing biomass that serve as one food base for an aquatic ecosystem; photosynthesis also provides an internal source of oxygen in the water body, in addition to oxygen from atmospheric exchange. Since algae produce oxygen, algal production affects day and night DO variations (diurnal) and summer DO variations (seasonal) in water column. During day time, algae increase DO while at night algae reduce DO through respiration, affecting diurnal DO variation. In summer particularly in late summer most of the organic matter derived from algae settles out from the water column where its decomposition contributes to oxygen depletion, influencing seasonal DO changes.

Algal biomass constitutes the amount of a living plant material within a specific volume or area in an aquatic ecosystem. In practice, total algal biomass is often represented by chlorophyll *a*, which is an indirect measure of the overall algae community. Since algal growth depends upon the conditions of light, water temperature and nutrients understanding the correlations between nutrients, algal biomass, and light transparency is vital for managing eutrophic lakes.

Figure 4.19 shows variation of chlorophyll *a* concentration at site A from 2010 to 2012. In 2010, average chlorophyll *a* concentration at site A throughout the water column from July and August varied between 7.5 and 54 µg/L; however, since the water column was vertically mixed in 2010 by aeration and phytoplankton were homogenously distributed (Figure 4.19a). In 2011, however, the lake was non-aerated and chlorophyll *a* at site A varied with depth (Figure 4.19b). For instance, at the surface and Secchi depth chlorophyll *a* concentration increased gradually from 30 to 90 µg/L by the end of July, then dropped during August to about 50 µg/L by the end of non-aerated period (Figure 4.19b). In addition, during non-aerated period chlorophyll *a* concentration at 1.5 m from the bottom layer (between 6 and 7 m depth) was less

compared to the chlorophyll *a* concentration at the surface and Secchi depth (Figure 4.19b). Furthermore, in 2011, results indicated that phytoplankton were not able to maintain their growth under non-aerated conditions because the phytoplankton population rapidly declined when nutrients were limiting in 2011. In comparison, in 2012, chlorophyll *a* concentration at site A was lower than in 2010 ranging between 7 and 20 µg/L (Figure 4.19c).

Dissolved inorganic nitrogen and dissolved inorganic phosphorous results indicated that nutrients accumulated in near-bottom layers during the non-aerated period (Figures 4.27 and 4.35), and these were not available for algae in the epilimnion; therefore, under non-aerated conditions in 2011, nutrients were limited for phytoplankton growth. Further, chlorophyll *a* results indicated decreased algal biomass due to lack of nutrients and competition during the stratified period of the lake (Figure 4.19b).

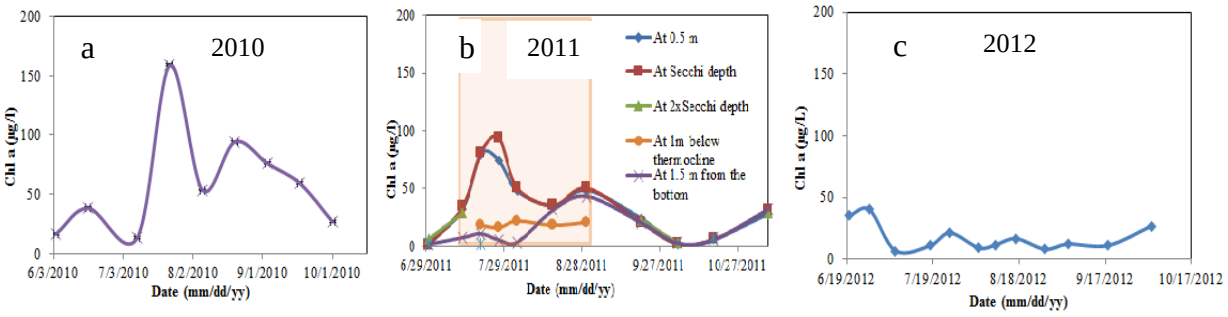

Figure 4.19: Chlorophyll *a* at site A in 2010, 2011 and 2012. The period without aeration is indicated by a square. In 2010 shows average chlorophyll concentration at surface and Secchi depth, and in 2012 shows chlorophyll concentration at Secchi depth

Similar to site A, site B, in 2010, showed chlorophyll *a* was approximately 7 and 60 µg/L (Figure 4.20a), and in 2011, the surface and Secchi depth was relatively higher than bottom layers, which increased from 20 to 90 µg/L during non-aeration (Figure 4.20b); however, in 2012, chlorophyll *a* at site B was similar to site A (Figure 4.20c). Shallow area sites C and D are shown in Figures 4.21 and 4.22. In 2010, at site C, chlorophyll *a* was between 28 and 100 µg/L (Figure 4.21a) and that was higher than sites A and B. In 2011, chlorophyll *a* showed

concentration gradient across the water column under non-aerated conditions (Figure 4.21b). In 2012, chlorophyll *a* concentration at site C was between 10 and 34 µg/L (Figure 4.21c). In contrast to sites A, B, and C, at site D chlorophyll *a* was quit high (ranged between 13 and 160 µg/L at the surface and Secchi depth) in 2010 (Figure 4.22a), while in 2011, chlorophyll *a* concentration was similar to sites A, B, and C, and algal biomass were peaked at the surface and Secchi depth compared to near bottom layers (Figure 4.22b). No significant differences were observed across sampling locations across the water column in 2010 and 2012 under aerated conditions; however, in 2011, under non-aerated conditions, chlorophyll *a* showed depth variability at all four sampling locations (Figures 4.19-4.22). In addition to sites A-D, in 2012, two new sites G and H were added and chlorophyll *a* concentration was between 8 and 40 µg/L (Figure 4.23).

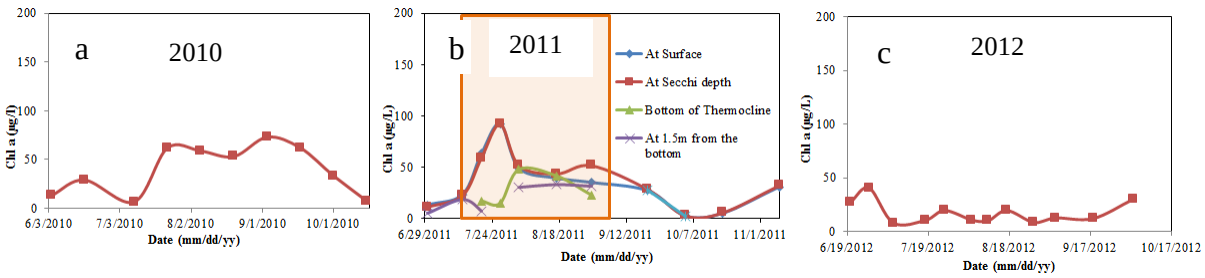

Figure 4.20: Chlorophyll *a* at site B in 2010, 2011 and 2012. The period without aeration is indicated by a square. In 2010 shows average chlorophyll concentration at surface and Secchi depth, and in 2012 shows chlorophyll concentration at Secchi depth

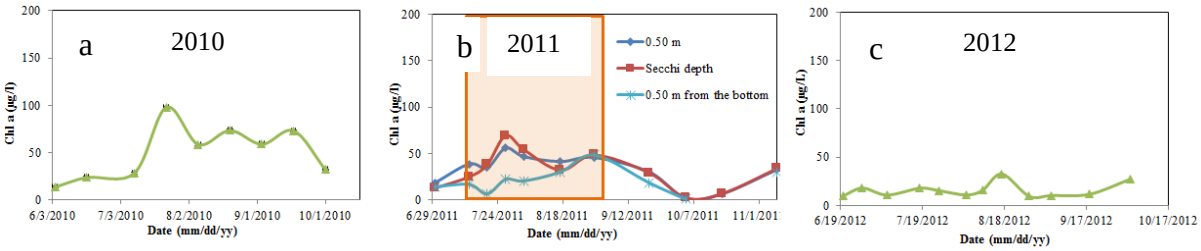

Figure 4.21: Chlorophyll *a* at site C in 2010, 2011 and 2012. The period without aeration is indicated by a square. In 2010 shows average chlorophyll concentration at surface and Secchi depth, and in 2012 shows chlorophyll concentration at Secchi depth

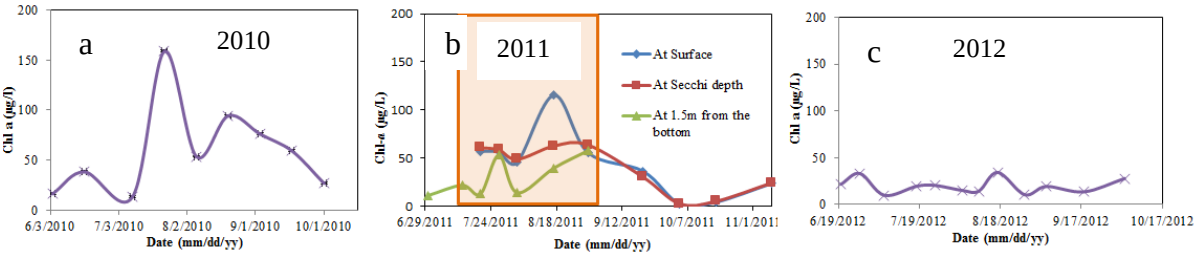

Figure 4.22: Chlorophyll *a* at site D in 2010, 2011 and 2012. The period without aeration is indicated by a square. In 2010 shows average chlorophyll concentration at surface and Secchi depth, and in 2012 shows chlorophyll concentration at Secchi depth

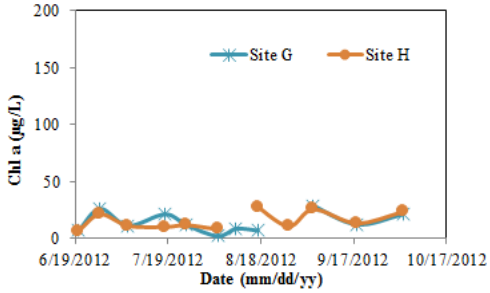

Figure 4.23: Chlorophyll *a* at surface in site G and H in 2012

The mean chlorophyll *a* concentration at the surface and Secchi depth in 2010 was higher than chlorophyll *a* concentration in 2011, chlorophyll *a* concentration under non-aerated conditions at all locations; however, only site A showed a significant increase. In addition, the mean chlorophyll *a* concentration in 2012, was significantly ($p<0.05$) lower at all sampling locations compared to 2011. Overall, in 2012, mean chlorophyll *a* concentration was

significantly ($p<0.05$) lower at the surface and Secchi depth at sites A, B, C and D. Therefore, artificial aeration decreased the mean chlorophyll *a* concentration significantly ($p<0.05$) from 2010 to 2012 at all sampling locations.

Wax et al., (2011) collected water quality data from reservoirs and impoundments in the rangeland plains ecological region of North Dakota between 1991 and 2011 and found that regional average chlorophyll *a* for 76 reservoirs was 19. 6 µg/L. In addition, NDDoH report that total maximum daily load (TMDL) for chlorophyll *a* for water quality in North Dakota is 20 µg/L. When compared to the current data, mean chlorophyll *a* concentrations at the surface and Secchi depth in July and August at all four sampling locations under both aerated and non-aerated conditions in 2010 and 2011 were greater than both TMDL and the regional average; however, in 2012, mean chlorophyll *a* concentration in July and August was between 12 and 17 µg/L at all sampling locations under aerated conditions. Furthermore, in 2010, at deep locations, mean chlorophyll *a* concentrations (35 and 38 µg/L) were less than non-aerated conditions (50 and 58 µg/L) in 2011; however, in 2010, under aerated conditions, at shallow locations, mean chlorophyll *a* concentrations (65 and 80 µg/L) were higher than non-aerated conditions (44 and 65 µg/L). In comparison, in 2012, chlorophyll *a* concentration was less than TMDL in both 2010 and 2011, which implies that from 2010 to 2012, lake water quality improved in terms of total algal biomass at the HMDI.

4.5. Water Clarity under Aerated versus Non-aerated Conditions

Water clarity is an indicator of water quality related to chemical and physical water properties and often indicates a lake's overall health. The maximum depth which algae can growth determines by light and euphotic zone is the upper portion, which light is sufficient for photosynthesis. The Secchi depth, turbidity and total suspended solids concentrations were

measured under aerated and non-aerated conditions from 2010 to 2012 as measures of water clarity.

4.5.1. Secchi Depth

The results of this study show the mean Secchi depth at sites A, B, C and D (1.6, 1.5, 1.2, and 1.25 m, respectively) under aerated conditions in 2010. Under non-aerated conditions, in 2011 sites A, B, C and D were 1.1, 1.2, 1.2 and 1.2 m, respectively. In contrast, in 2012, Secchi depth was greater than 2 m at all sampling locations under aerated conditions (Figure 4.24). These results indicate that under aerated conditions in 2012, lake water quality improved to the level of EPA standards for lakes and reservoirs (2 m) (EPA, 2001).

The mean Secchi depth decreased from 2010 to 2011 but not significantly at all sampling locations; however, from 2011 to 2012, the mean Secchi depth increased significantly ($p<0.05$) at all locations. Comparing the mean Secchi depth from 2010 to 2012, results indicated that the HMDI water quality improved by the artificial aeration system, and the mean Secchi depth was significantly higher in 2012 than in 2010.

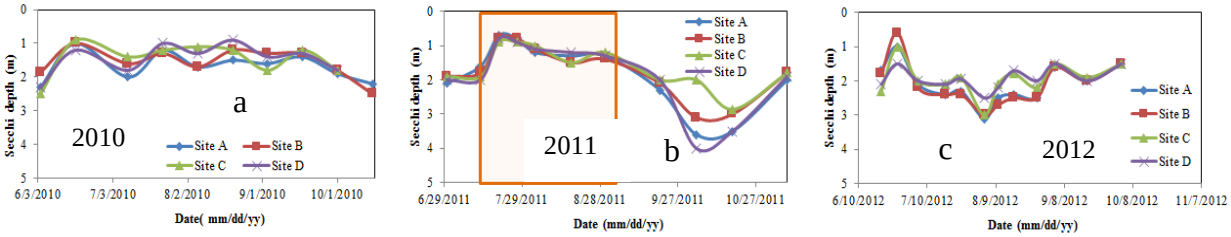

Figure 4.24: Variations of Secchi depth in 2010, 2011 and 2012. The period without aeration is indicated by a square

Secchi depth values vary as algal populations increase and decrease. Comparing chlorophyll *a* concentrations with Secchi depth in Figure 4. 25, it is evident that there is an inverse relationship between Secchi depth and chlorophyll *a* concentration in the water column. The Secchi depth is an indicator of water clarity; however, it is not always an indicator of trophic state, as in those waterbodies in which the water clarity is controlled by some other factors, such

as variable amounts of chlorophyll among algal species, variable amounts of chlorophyll within an algal species as a function of cell age and nutrition, contribution of other suspended particles , the volume versus number of phytoplankton cells, and variable chlorophyll extraction efficiency as a function of cell age and solvents used (Lind, 1986; Lee et al., 1995; Mazumder and Havens, 1998).

Among those factors, most likely the main factor affecting the relationship between chlorophyll *a* and Secchi depth at the HMDI was volume versus number of phytoplankton cells because based on lake water quality conditions at HMDI the dominant phytoplankton genera in summer were *Aphanizomenon* sp, *Microcystis* sp and *Anabaena* sp. The number of cells per volume in each genus are different due to their different morphological features , such as *Aphanizomenon s*p. are filamentous and united to form plate-like bundles and flakes of parallel trichomes while *Microcystis* are sedentary colony of numerous spherical cells in copious mucilage forming globes, and affecting for the Secchi depth turbidity (Prescott, 1982).

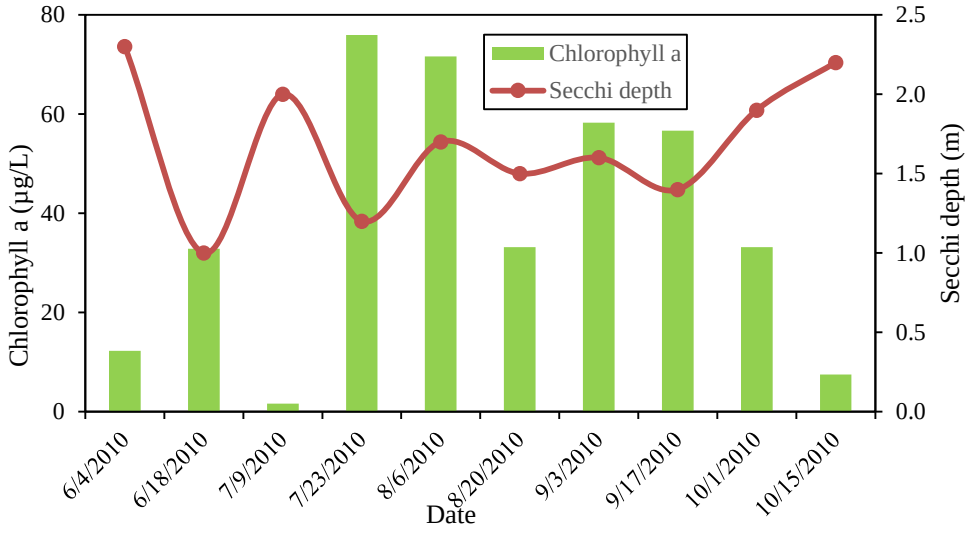

Figure 4.25: Variation of chlorophyll *a* at Secchi depth level and lake's Secchi depth at site A in 2010

In addition to Secchi depth, other measures of water clarity are turbidity and total suspended solids. Suspended particles dissipate light, which affects the depth at which algae can grow and total suspended solids (TSS) include all suspended particles, such as phytoplankton, zooplankton, decaying plant and animal matter, and silt in the water column. Figure 4.26 shows Secchi depth varied in response to TSS concentration as seen in Figure 4.25. Figures 4.25 and 4.26 show that Secchi depth varied not only because of phytoplankton biomass but also no-algal turbidity in the water column. Turbidity also affects the aesthetic quality of water, and algae growth is a common reason for low Secchi depth.

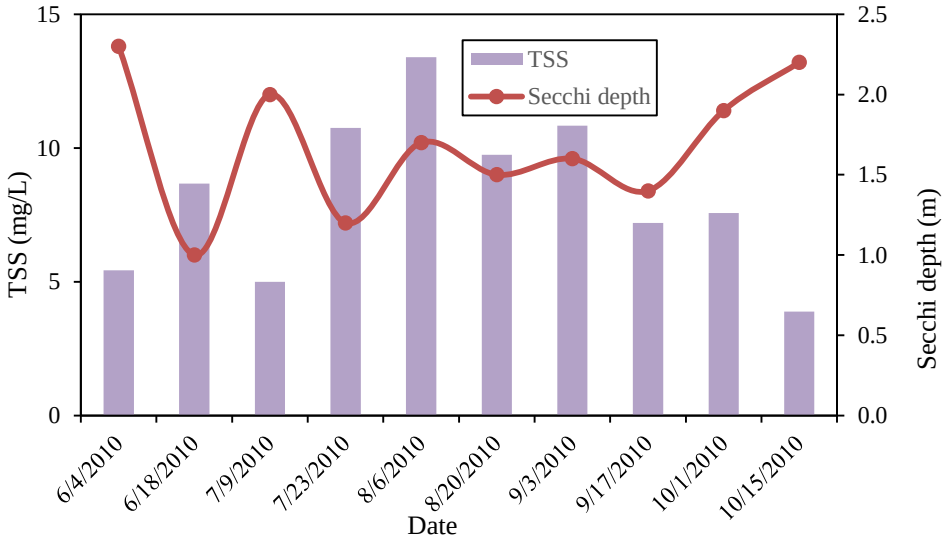

Figure 4.26: Variation of total suspended solids at Secchi depth level and lake's Secchi depth at site A in 2010

4.5.2. Turbidity and Total Suspended Solids (TSS)

The mean turbidity at sites A,B,C and D, respectively, between July and August was 4.3, 4.5, 4.8, and 5.1 NTU at the surface and Secchi depth in 2010; however, turbidity was 8.4, 8.5, 7.5 and 7 NTU at the surface and Secchi depth at sites A,B,C and D, respectively, under the non-aerated period in 2011. In comparison, in 2011, turbidity highly depended upon the contribution of phytoplankton at the surface and Secchi depth; however, in 2012, the mean turbidity at the surface and Secchi depth dropped to less than the turbidity in 2010 and was approximately 2 NTU. The results imply that lake water quality improved in 2012 under the conditions of aeration at the HMDI. The mean turbidity increased significantly at the surface and Secchi depth at sites A, B, and C from 2010 to 2011; however, site D showed no significant differences in turbidity between those two years. Furthermore, the mean turbidity decreased significantly ($p<0.05$) at all four locations from 2011 to 2012 as well as the mean turbidity decreased

significantly ($p<0.05$) at all locations from 2010 to 2012. The results indicated that overall water quality improved from 2010 to 2012 at the HMDI during the study period.

In addition to turbidity, measurements of TSS also showed that the variation of light transparency from 2010 to 2012 and often depended upon contribution of phytoplankton. In 2010, the mean TSS at deep location sites A and B at the surface was 7.5 and 7.8 mg/L, while in 2011, TSS was between 10 and 11 mg/L at sites A and B; however, in 2010 and 2011, TSS at shallow location sites C and D were between 10 and 12 mg/L. Total suspended solids were not measured in 2012 at any location; therefore, in comparison, under the non-aerated conditions, TSS were higher than the aerated conditions in 2010, at deep locations and TSS were similar in both 2010 and 2011 at shallow locations. In addition, TSS were significantly ($P<0.05$) greater at the surface layer than the bottom layer during the non-aerated conditions at sites A, B, and C, while no differences were observed at site D. During the non-aerated conditions, the majority of phytoplankton were reported at the surface and Secchi depth and the majority of TSS were contributed by phytoplankton at the surface and Secchi depth; therefore, significantly high phytoplankton were reported at the surface water column than the bottom layers. The EPA recommended value for suspended solids is 30 mg/L; therefore, the TSS of this study was less than the EPA recommended value in all three years regardless of the aeration at all sampling locations.

4.6. Variation of Nutrients under Aerated versus Non-aerated Conditions

Nutrients are necessary for the growth of living organisms, including algae, however excessive nutrients can be harmful to the ecosystem. The two nutrients of greatest concern are nitrogen and phosphorous. A previous study (Overmoe, 2008) carried out at the Heinrich Martin Dam Impoundment (HMDI) observed both total nitrogen (TN) and total phosphorous (TP)

concentrations steadily increase in the water column during the algae growing season (from the end of May to the end of August) under the aerated conditions. Total nitrogen concentration was greater than 0.15 mg/L, while TP concentration was greater than 0.01 mg/L (Overmoe, 2008), and both concentrations exceeded the minimum concentrations, which predict nuisance algal blooms (Gibson et al. 2010). Since the artificial aeration is a costly method, understanding changes of main nutrients in the water column under the aerated conditions is vital for eutrophic lake management. The previous research, however did not measure changes in algal biomass and nutrient variation under artificial aerated conditions.

The research described here was carried out in 2010, 2011 and 2012 aimed to decide whether existing artificial aeration system improve lake water quality by measuring the variation of nutrient concentration under aerated versus non-aerated conditions and its influence on algae population and other physical and chemical water quality parameters (water temperature, dissolved oxygen, conductivity, pH, and light transparency).

In 2010, sampling was carried out under the aerated conditions from June to October, and in 2011, sampling was carried out under the non-aerated conditions from July to August. In 2012, sampling was carried out under the aerated conditions from June to October. In three years, therefore, sampling was carried out approximately from June to October, which is the algae growing season, and on-site field measurements and samples were collected at four sampling locations (sites A, B, C and D). Results in 2010, 2011 and 2012 under the aerated and the non-aerated conditions are reported here.

4.6.1. Nitrogen

The dissolved inorganic forms of nitrogen that are taken up by algae and plants includes ammonia-N (NH_3-N), nitrite-N (NO_2-N) and nitrate-N (NO_3-N). In addition to dissolved

inorganic nitrogen, organic nitrogen and TN in the water column are also important to predict effective lake management strategies. Organic nitrogen undergoes decomposition and is mineralized to ammonia to return to the water column.

In 2011 and 2012 water samples for nutrients were collected at all four sites, while in 2010, samples for nutrient were collected only at site B because spatial variation of nutrients was not expected. In 2011, samples were collected across the water column to determine nutrient concentration variation under non-aerated conditions; however, in 2012, samples for nutrient analyses were only collected at the surface layer assuming homogenous distribution of nutrients across the water column under aerated conditions. In 2011, NH_3-N concentration at Secchi depth and 2× Secchi depth was zero, while NH_3-N at near bottom layers was between 0.1 and 1 mg/L during the non-aerated period at site A (Figure 4.27). In comparison, in 2012, NH_3-N concentration was between 0.1 and 0.2 mg/L at the surface from the end of July to the end of August at site A (Figure 4.27). Similar to site A, at site B, NH_3-N concentration was low in all three years 1) in 2010, when aeration was operational, NH_3-N varied from 0.05 to 0.12 mg/L (Figure 4.28), 2) in 2011, NH_3-N at the surface and Secchi depth was zero, while concentrations slightly increased to 1 mg/L at near- bottom layers at the beginning of the sampling (Figure 4.28), and 3) in 2012, NH_3-N concentration at surface at site B was similar to site A and ranged between 0.1 and 0.2 mg/L (Figure 4.28). In addition to deep locations, NH_3-N was measured at shallow location sites C and D, which were located far from the aeration system to assess influence of aeration on the whole water body. These sites, C and D, are shallow; therefore, samples for nutrient analyses were only carried out at the surface or Secchi depth level. As shown in Figures 4.29 and 4.30, NH_3-N concentration was zero at both sites C and D under non-aerated conditions, while under aerated conditions in 2012, concentrations varied between 0.02

and 0.2 mg/L at both sites. Results indicated NH_3-N was completely exhausted by algae for their growth at surface and Secchi depth. Under non-aerated conditions, stratification prevented homogeneous mixing of nutrients across the water column; therefore, nutrients in near bottom layers were not directly available for algae in surface layers.

The mean ammonia-N concentration at the near bottom layers compared to the surface water column at deep locations decreased significantly under aeration in 2010; however, mean ammonia-N concentrations at the surface layers from 2011 to 2012 increased significantly ($p<0.05$) at all sampling locations. In productive lakes, ammonia commonly accumulates in bottom waters in conjunction with the onset of anoxic conditions due to a loss of biological nitrification; however, under oxic conditions ammonia does not accumulate and is oxidized to nitrate. According to the EPA's criteria for water quality, chronic toxicity for fish species ranges from 0.0017 to 0.612 mg/L NH_3 and acute toxicity to salmonids ranged from 0.083 to 1.09 mg/L NH_3 and from 0.14 to 4.60 mg/L NH_3 for non-salmonids. In this study, the mean NH_3-N concentration was greater than the EPA's minimum concentration of 0.0017 mg/L at both the surface and the bottom under aerated and non-aerated conditions; mean ammonia-N was closer to the maximum chronic toxicity value (0.612 mg/L NH_3) in near bottom layers under non-aerated conditions. Since the HMDI is managed as a bluegill fishery, and bluegill from the HMDI are used for brood stock around the state, ammonia-N concentration needs to be maintained within the acceptable range. Under the non-aerated conditions in 2011, summer fish kill was observed and was possibly due to ammonia toxicity.

Nitrite-N concentration was zero due to the inhibition of nitrification because of low DO at the bottom layer during the study period in 2010, 2011 and 2012 under aerated and non-aerated conditions at all sampling locations. Nitrate-N concentration in 2010 was approximately

0.1 mg/L, and in 2011, NO_3-N concentrations during non-aerated period at all four locations were between 0.05 and 0.09 mg/L and no differences were observed in near bottom layers at sites A and B as compare to significance difference observed for NH_3-N in Figures 4.27 and 4.28. In 2012, NO_3-N at all four locations was between 0.12 and 0.25 mg/L during the aerated period.

Results of this study show that mean nitrate and nitrite concentrations significantly ($p<0.05$) decreased from 2010 to 2011 at surface at site B (data not available at sites A, C, and D in 2010) and significantly ($p<0.05$) increased from 2011 to 2012 at surface layers at sites A, B and D; however, no significant differences were observed at site C. Overall, the mean nitrate and nitrite concentration increased significantly ($p<0.05$) under aeration. The nitrite-N concentration was zero at all sampling depths at all locations during the study period. Furthermore, the limiting nitrate concentration goal for lake and reservoir improvement or maintenance in North Dakota is 0.25 mg/L. In this study, NO_3-N concentration increased under aeration and ranged from 0.11 to 0.16 mg/L at all sampling locations; however, NO_3-N concentration was less than the goal value. Results suggested that the aeration does influence variation in NO_3-N but does not exceed the goal concentration.

Results for all three years indicated that dissolved inorganic nitrogen (DIN), comprised of NH_3-N, NO_3-N and NO_2-N, was below the limiting concentration (0.4 mg N L-1) for phytoplankton growth at surface / Secchi depth level (Phlips et al., 1997; Havens et al., 2003) under both aerated and non-aerated conditions. Ammonia-N concentration in near bottom layers increased to 1 mg/L during the non-aerated period only at deep locations. Under oxic conditions, NH_3-N could further oxidize into NO_3-N; therefore, NH_3-N did not accumulate in near bottom layers under aerated conditions in 2010 and 2012; however, under non-aerated period, due to

anaerobic conditions in near-bottom layers, NH_3-N may not further oxidize into NO_3-N; therefore, NH_3-N accumulated near the bottom. Ammonia-N, which accumulated near the bottom was not available for algae at the epilimnion (the top-most warm well mixed layer of a stratified lake) region; therefore, during the non-aerated period in 2011, there was a nitrogen limiting condition for algal growth in the epilimnion. In contrast, under aerated conditions in 2010 and 2012, nutrients homogenously mixed across the water column due to artificial aeration. Compared to the aeration versus non-aeration conditions in 2010, 2011 and 2012, NH3-N was zero at the euphotic zone where algal growth is possible and expected at all locations under the non-aerated period. In contrast, NH_3-N was slightly high at the surface and Secchi depth under aerated conditions because there was a supply of nutrients to the surface, which algae could use for their growth.

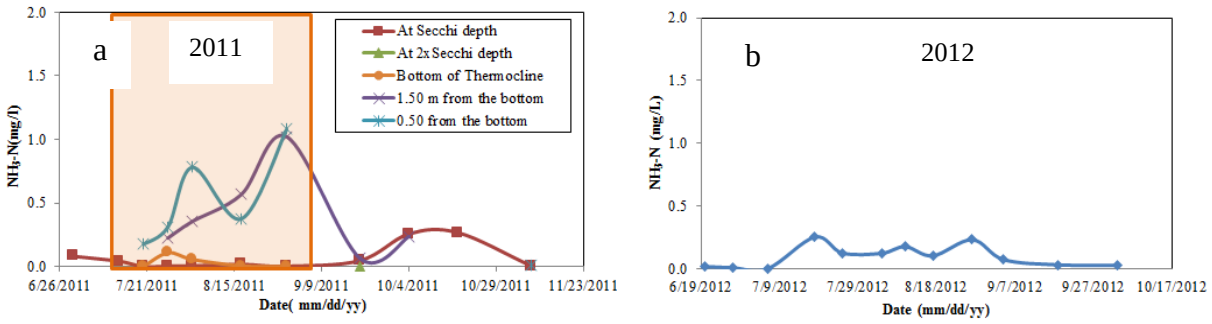

Figure 4.27: Variations of NH_3-N at site A in 2011 and 2012. In 2012, NH_3-N measured only at Secchi depth. The period without aeration is indicated by a shaded area

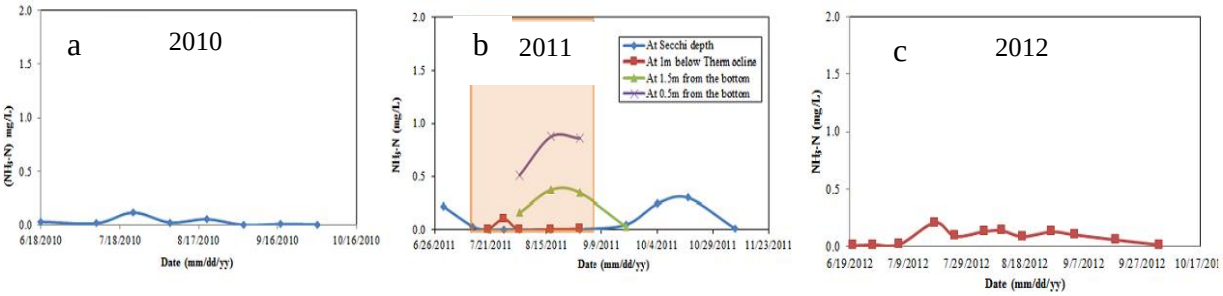

Figure 4.28: Variations of NH_3-N at site B in 2010, 2011 and 2012. In 2010 and 2012, NH_3-N measured only at Secchi depth. The period without aeration is indicated by a shaded area

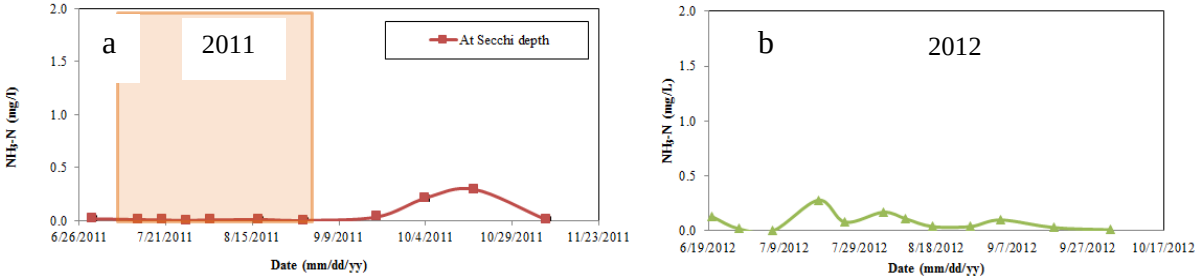

Figure 4.29: Variations of NH_3-N at site C at Secchi depth in 2011 and 2012. The period without aeration is indicated by a shaded area

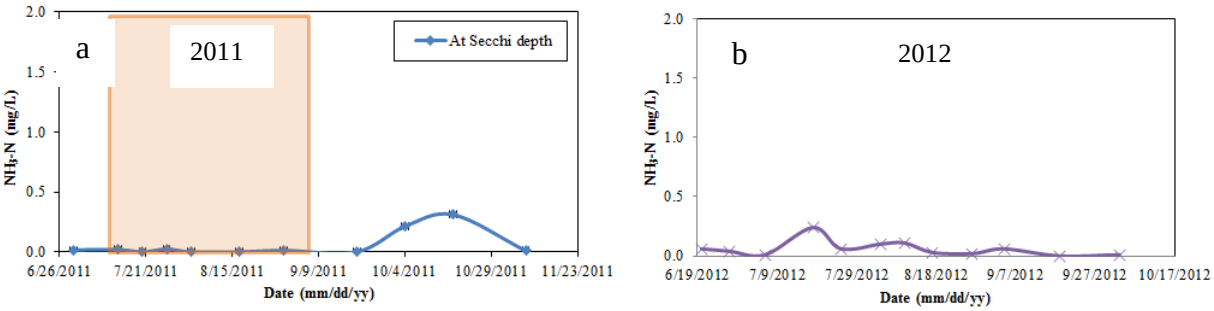

Figure 4.30: Variations of NH_3-N at site D at Secchi depth in 2011 and 2012. The period without aeration is indicated by a shaded area

In addition to inorganic nitrogen, TN was also measured in all three year to determine trophic status of the water body under aerated and non-aerated period. Total nitrogen concentration better reflects the trophic status of a water body, since algal growth significantly affects the dissolved inorganic nutrients. Algae directly utilize dissolved inorganic nitrogen and deplete it during algal growth; however, low dissolved nitrogen does not necessarily result in low

algal growth because TN concentration in the water column can increase due to transformation of organic nitrogen into inorganic nitrogen.

Figure 4.31 shows variation of TN at site A in 2011 and 2012. In contrast to the variation of NH_3-N in 2011, TN at all sampling depths gradually increased at the beginning of the sampling; however, concentrations at all the depths gradually declined (Figure 4.31). Increased TN at the surface and Secchi depth could possibly be due to phytoplankton growth, and increased TN in near bottom layers is due to the accumulation of NH_3-N (Figure 4.27) and could possibly be due to organic sediments. In 2011, TN at site A ranged between 1 and 4 mg/L (Figure 4.31), and the majority of TN was represented by organic nitrogen since the total dissolved inorganic nitrogen was approximately 1 mg/L; therefore, 25% of TN was represented by inorganic nitrogen and 75% of TN was represented by organic nitrogen. In 2012, TN concentration varied between 2.5 and 3.5 mg/L at the surface layer at site A during the phytoplankton growing season that indicated the majority of TN was comprised by phytoplankton (Figure 4.31).

Total nitrogen concentration in lakes may change due to external run off, internal nutrient loading and changes of algal biomass. Figure 4.32 shows the variation of TN at site B. As shown in Figure 4.31, in 2010, TN varied from 1 to 1.5 mg/L between July and August. When compared to the same period in 2011 and 2012 at site B, TN was much higher in 2011under non-aerated condition (1 to 5 mg/L) at 1m below the bottom of the thermocline (transition zone between epliminion and hypolimnion), while TN at the rest of the depths were between 1 and 3.5 mg/L (Figure 4.32). Total nitrogen was significantly high at site B in 2012 (45 mg/L) and most probably due to impacts of precipitation. Similar to deep locations, at shallow locations, TN gradually increased at Secchi depth level in 2011 from 1 to 3 mg/L because algae accumulated at

the surface and Secchi depth during the non-aerated period and algal population dropped by the end of August (Figures 4.33 and 4.34). Total nitrogen concentration was 2 and 4 mg/L during the aerated period in 2012 at site C (Figure 4.33) due to algal biomass changes, while TN concentration was significantly high at only one sampling date (7/6/2012) at site D (Figure 4.34).

The mean TN concentration increased from 2010 to 2011 at site B but not significantly and the mean TN at surface increased from 2011 to 2012 at all locations; however, among those sites only at sites B and C showed a significant increase from non-aerated period to aerated period. The EPA recommended value for TN is 0.44 mg/L and regional average value for TN is 1.32 mg/L. The results of this study showed the mean TN concentration under both aerated and non-aerated conditions was greater (between 2 and 3 mg/L) than the standard value as well as the regional average of North Dakota reservoirs and impoundments.

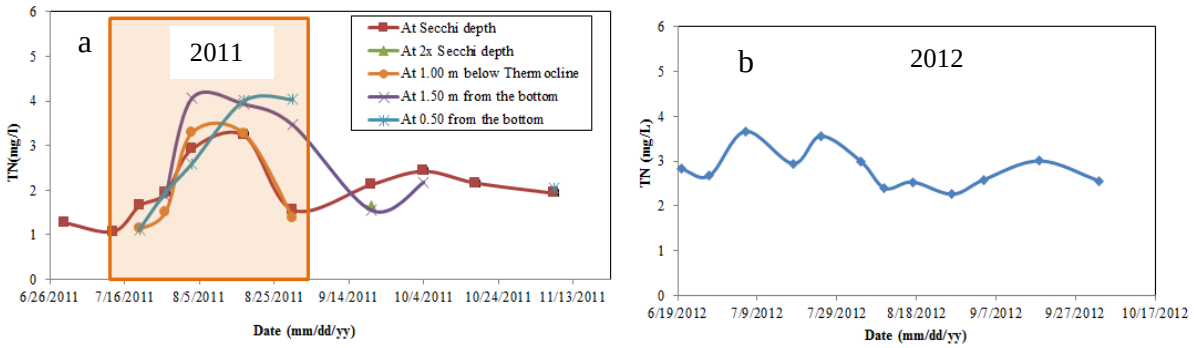

Figure 4.31: Variations of TN at site A in 2011 and 2012. In 2012, sampling depth was Secchi depth. The period without aeration is indicated by shaded areas

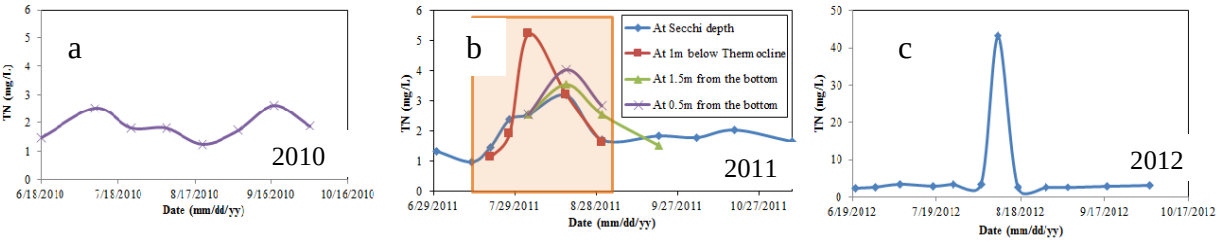

Figure 4.32: Variations of TN at site B in 2010, 2011 and 2012. In 2010 and 2012 sampling depth was Secchi depth. The period without aeration is indicated by shaded areas

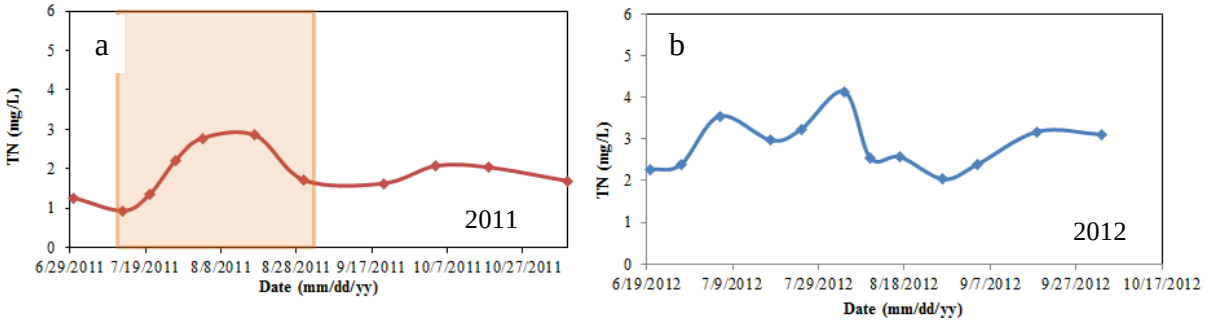

Figure 4.33: Variations of TN at site C at Secchi depth in 2011 and 2012. The period without aeration is indicated by shaded areas

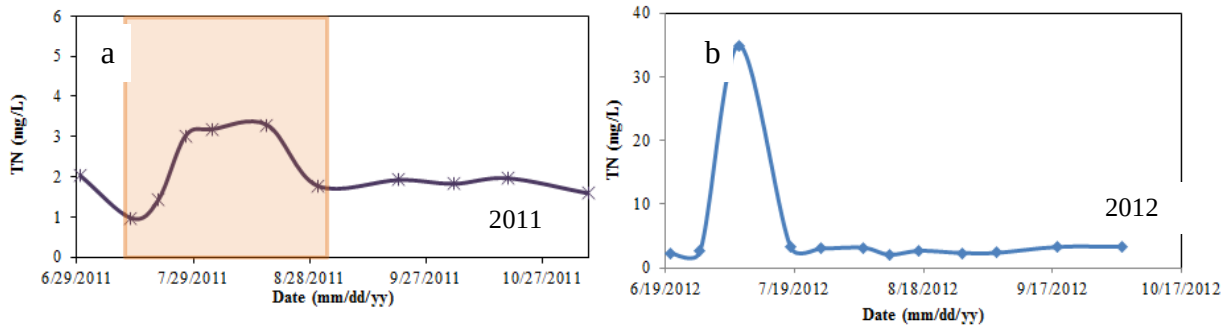

Figure 4.34: Variations of TN at site D at Secchi depth in 2011 and 2012. The period without aeration is indicated by shaded areas

In 2011 and 2012, in addition to TN, total dissolved nitrogen (TDN) was also measured to identify the composition of TN. Results indicated that the majority of TN was comprised by dissolved nitrogen either as organic or inorganic at all sampling locations. In 2011, at deep location sites A and B, TDN represented 63 to 98% of TN at Secchi depth and 90 to 97% of TN in near bottom layers. In addition, TDN at shallow location sites C and D, ranged between 70 and 98% at Secchi depth. In 2012, TDN was between 70 to 100% at Secchi depth in all sampling locations except on August 10th at site B (TDN was only 4% of TN). Results in both 2011 and 2012 showed that the majority of TN was in the form of TDN regardless of aeration at all sampling locations. Since dissolved inorganic nitrogen concentration was less than 1 mg/L at all

sampling locations, the majority of dissolved nitrogen was in the form of dissolved organic nitrogen.

Zehr et al (1988) found sediment release was the major source of dissolved organic nitrogen (DON) in meso-oligotrophic Castle Lake, California. In another study, DON was also found DON to be the dominant N form, which was released from sediments (Middleboe et al., 1998). In the current study, the high percentage of DON compared with DIN is due to the fact that the organic nitrogen seems to be assimilated by algae at a much slower rate than that of the inorganic nitrogen species. In general, the present data show that DON is an important source of nitrogen in the lake water.

4.6.2. Phosphorous

Phosphorous is another vital nutrient and often functions as the limiting nutrient in many freshwater systems and thereby determines phytoplankton growth; however, excessive amounts cause algal blooms and lead to eutrophication problems (Sondergaard, 2007). Phosphorous occurs in dissolved inorganic, organic or particulate forms, and the dissolved inorganic form, soluble reactive phosphorous (SRP), is the readily available form for algal uptake and plant growth; therefore, it is very important to determine SRP under aerated and non-aerated conditions to identify which lake conditions prevent algal blooms and water quality deterioration. In addition, unlike nitrogen, SRP settles with sediment solids to the bottom of the water body and later becomes a phosphorous source to the water body; therefore, bottom sediments contain enough phosphorous to accelerate eutrophication. Quantifying bioavailable phosphates under aerated versus non-aerated conditions would provide useful information to lake managers to prevent lake eutrophication.

Similar to TN, total phosphorous (TP) also better reflects the trophic status of a water body than SRP; therefore, quantifying TP under aerated versus non-aerated conditions is crucial information to prevent lake eutrophication.

4.6.2.1. Soluble Reactive Phosphorous

Figures 4.35 and 4.36 show the distribution of SRP at deep locations under aerated versus non-aerated conditions. The SRP concentration at Secchi depth level at deep locations was between 0.5 and 1 mg/L, while in the near bottom layers, SRP concentration increased to 0.45 mg/L in 2011(Figures 4.35 and 4.36). In comparison, in 2010, average SRP across the water column at site B gradually increased and reached a peak of 0.2 mg/L by mid-August (Figure 4.36), and in 2012, SRP at the surface layer also showed the same trend and reached a peak concentration of 0.2 mg/L by mid-August (Figure 4.36). Under non-aerated conditions, vertical mixing was prevented by lake stratification because the eplimnion is isolated from the bottom hypolimnion (colder, undisturbed bottom layer of a stratified layer). Nutrients were not homogenously mixed and bottom nutrients were not available for algal growth in the eplimnion; therefore, under non-aerated conditions, SRP at the surface and Secchi depth was low compared to the aerated conditions. Under aerated conditions due to vertical mixing, nutrients redistributed evenly across the water column and provided SRP for algal growth. Therefore, under aerated conditions, SRP was higher than for non-aerated conditions. Under aerated conditions in 2010, SRP was slightly high at the surface layers at both sites C and D (Figure 4.37) compared to the non-aerated conditions in 2011. There were no significant differences observed in SRP at sites C and D in both 2011 and 2012 (Figure 4.37).

Since SRP is released into the water column by aerobic or anaerobic decomposition, and Fe-desorption. The previous study (Overmoe, 2008) carried out at the HMDI showed that total

iron concentration in the water column was under the detectable level; therefore, SRP release by Fe-desorption under anaerobic conditions was insignificant. A possible mechanism of SRP release under non-aerated conditions is anaerobic decomposition, while under aerated conditions, SRP is likely released by aerobic decomposition. In comparison, aeration produced a 3-4 % increase in SRP at the surface and Secchi depth in shallow areas, while in deep areas, aeration produced a 6-8 % increase in SRP at the surface and Secchi depth.

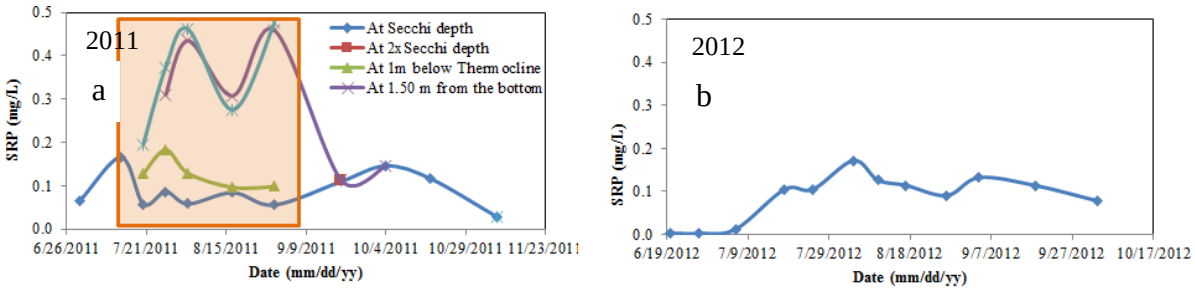

Figure 4.35: Variation of SRP at site A in 2011 and 2012. In 2012 sampling depth was Secchi depth. The period without aeration is indicated by a square

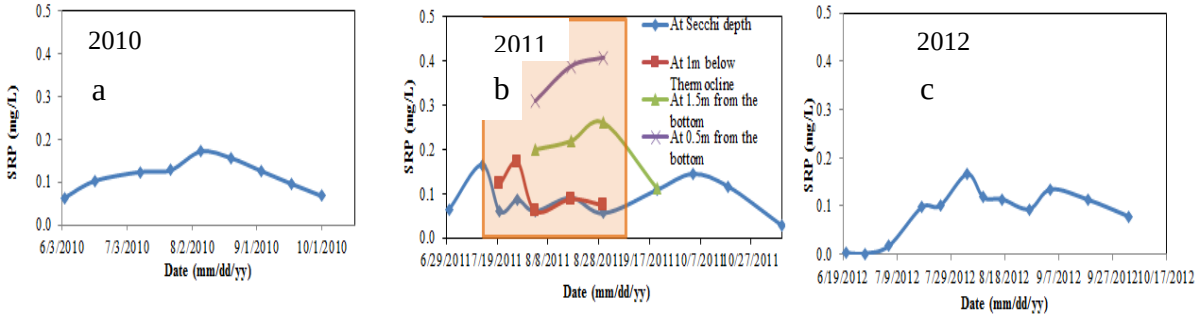

Figure 4.36: Variation of SRP at site B in 2010, 2011 and 2012. In 2010 and 2012 sampling depth was Secchi depth. The period without aeration is indicated by a square

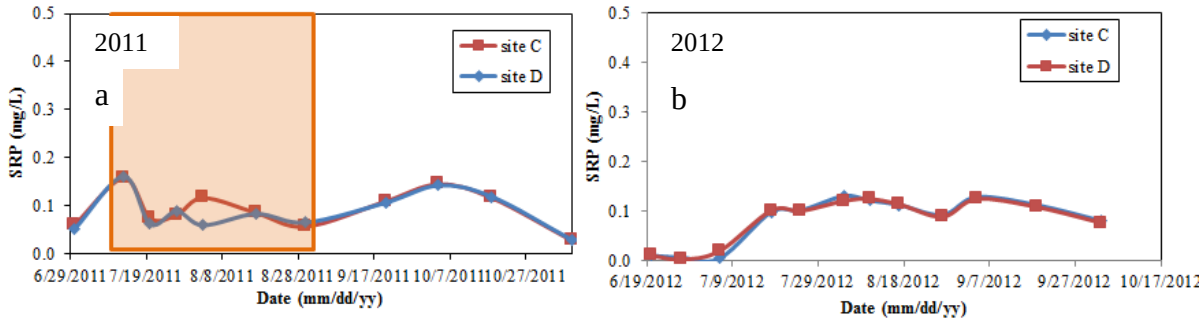

Figure 4.37: Variation of SRP at sites C and D at Secchi depth in 2011 and 2012. The period without aeration is indicated by a square

The mean SRP concentration at surface water column increased but not significantly from 2011 to 2012 at all sampling locations. The recommended standard value for lake or reservoir improvement for North Dakota is 0.02 mg/L and the mean concentration was greater than both aerated and non-aerated conditions. Therefore, aeration did not meet the North Dakota Department of Health (NDDoH) orthophosphates concentration for lake and reservoir.

4.6.2.2. Total Phosphorous

Figure 4.38 shows the influence of aeration on TP at site A. In 2011, at the Secchi depth and surface, TP varied from 0.22 to 0.30 mg/L (Figure 4.38), and in 2012, TP increased under aerated conditions and ranged between 0.30 and 0.44 mg/L at the surface (Figure 4.38). In near bottom layers, in 2011, TP concentration was higher than at the surface (varied between 0.3 and 0.7 mg/L). In 2011, non-aerated conditions, bioavailable dissolved inorganic phosphorous at site A was depleted as shown in Figure 4.35, and nutrients accumulated in near bottom layers due to lake stratification. Similar to site A, Figure 4.39 shows that at site B in 2011, TP accumulated in near bottom layers (between 0.2 and 0. 56 mg/L) and that TP concentration was higher than at the surface and Secchi depth (ranged between 0.17 and 0.34 mg/L) (Figure 4.39). Under aerated conditions in 2010 and 2012, TP was between 0.17 and 0.25 mg/L at the surface and Secchi depth and approximately the same as TP at the surface and Secchi depth under non-aerated

conditions. As shown in Figure 4.40, for sites C and D, TP at the surface and Secchi depth in shallow locations initially decreased but them increased by the latter part of the non-aerated conditions due to increase of algal biomass in 2011 (Figure 4.40). Compared to the non-aerated period in 2011 (between July and August), in 2012, TP at sites C and D increased to 0.5 mg/L and was higher than non-aerated conditions in 2011 (Figure 4.40).

In comparison, the mean TP increased at surface water column at all locations from 2011 to 2012; no significant increase was observed at any sampling locations; however, only significant differences in TP were observed between surface and bottom in 2011. The mean TP concentration was greater than the regional average (0.126 mg/L) of other reservoirs and impoundments in the North Dakota.

In 2010, approximately 60-74 % of TP was in the form of SRP in the water column, indicating rapid decomposition of organic phosphorous in the sediment; however, in 2011, around 18-44% of TP was in the form of SRP at the surface and Secchi depths and the majority of TP was in the form of SRP in near bottom layers and was not readily available to the epilimnion under non-aerated conditions. Under aerated conditions, since a majority of SRP distributed evenly across the water column by vertical mixing, more habitats for phytoplankton were available, which could eventually lead to excessive growth and lake eutrophication.

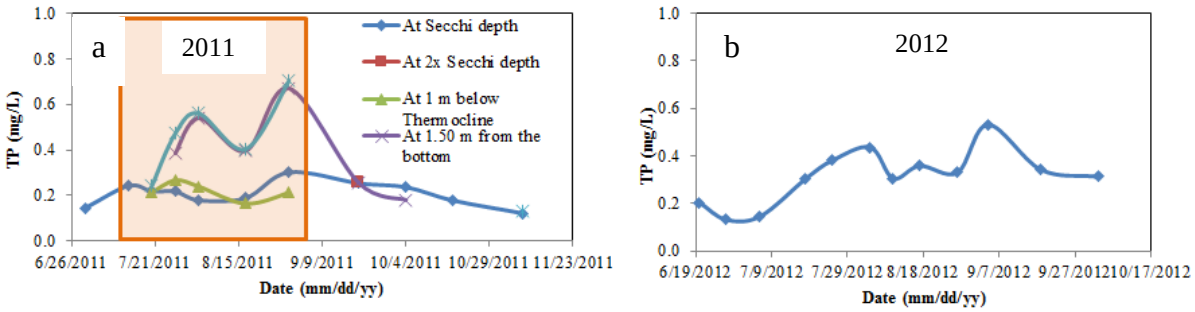

Figure 4.38: Variations of TP concentration at site A in 2011 and 2012. In 2012 sampling depth was Secchi depth. The period without aeration is indicated by a square

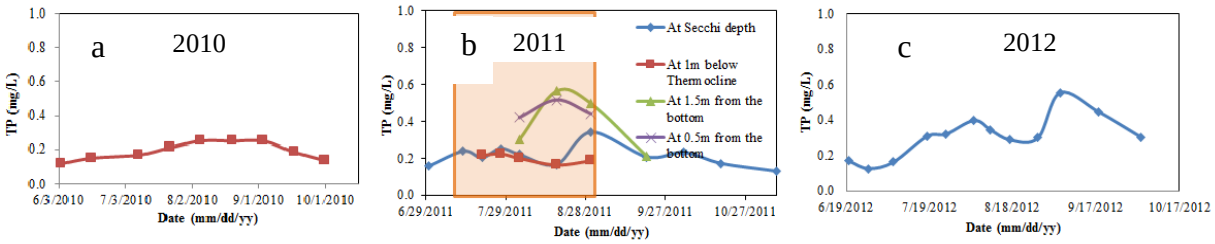

Figure 4.39: Variations of TP at site B and precipitation in 2010, 2011 and 2012. In 2010 and 2012 sampling depth was Secchi depth. The period without aeration is indicated by a square

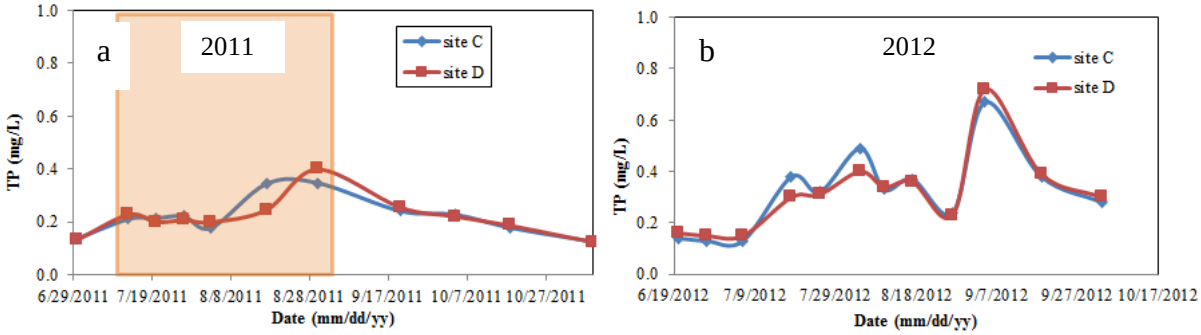

Figure 4.40: Variations of TP at Secchi depth in sites C and D in 2011 and 2012. The period without aeration is indicated by a square

4.6.3. N: P Ratio

The dissolved nitrogen-to-dissolved phosphorous ratio is used as a rough estimation to assess which nutrient is limiting plant growth in lakes. The ratio of dissolved nitrogen to dissolved phosphorous in biomass is approximately 7.2:1; therefore, a N: P ratio in water that is less than 7.2:1 indicates dissolved nitrogen is limiting. In contrast, when the ratio is higher than 7.2:1, dissolved phosphorous is limiting algal growth. In 2010, the ratio of DIN to SRP at site B was between 1 and 2 at the beginning of the sampling and dropped to 0.8 as the season progressed. By the end of sampling, the DIN: SRP ratio reached 3. A similar trend was observed in 2011. The DIN: SRP ratio in HMDI was between 6 and 9, which was closer to the Redfield ratio and dropped to 0.8 as the season progressed. The DIN: SRP ratio reached 4 by November at all sampling locations. Further, in 2012, the DIN: SRP at HMDI was also less than

Redfield ratio. Therefore, during the study period results showed the DIN: SRP ratio was less than 7.2:1, which indicates the HMDI is N-limiting, and there is an excess supply of available P in the water column.

4.7. Discussion

The aim of this study was to determine whether the aeration system used in HMDI improved lake water quality by comparing chemical and physical water quality parameters under the conditions of aeration and non-aeration. Artificial aeration is the only management strategy used in HMDI; the aerator was installed in May, 2006, and operation began in May/June each year in summer for the purpose of improving DO in the near bottom layers. The aerator was in operation from 2006 to 2008, but not during 2009 because of high snowfall during winter 2008-2009. According to the North Dakota Game and Fish Department, when the impoundment reaches its maximum flow level, the aeration system is shut off and the hypolimnetic discharge system is turned on to remove excess water. In addition to artificial aeration, a hypolimnetic siphon is operated annually used in the HMDI to control water level.

The artificial aeration of waters, like in most of the lakes restored using this method, (Cooke et al. 2005; Grochowska and Gawronska, 2004) can cause a substantial decrease of TP; however, other studies indicate that aeration has no lasting effect on the phosphorous concentration in aerated lakes (Ashley, 1983; McQueen et al. 1984; Gachter and Wehrli, 1998). The current study, although SRP and TP increased under aerated conditions at the HMDI, this increase was not statistically significant. Before the aeration system was installed at the HMDI (between 1992 and 1993), the mean SRP was 0.038 mg/L; the recommended value for lakes and reservoir improvements by NDDoH is 0.02 mg/L. In comparison, the SRP value of this study was between 0.08 and 0.1 mg/L throughout the study period. Therefore, the HMDI water quality

did not improve as needed by NDDoH requirements (NDDoH, 2001). The increase in SRP was probably due to enhancement of organic matter decomposition in the sediments under improved DO concentrations in the near bottom layers (by the artificial aeration) and non-point source pollution. In addition to SRP, the historical mean for TP (1992-2005) in the HMDI is 0.087 mg/L; the results of this study showed the mean TP at surface to be 0.32 mg/L by the end of the study, which indicated that during the study, TP increased. The EPA recommended value for TP in lakes and reservoirs in nutrient ecoregion IV is 0.02 mg/L, and TP load needs to be lowered according to the EPA requirements (EPA, 2010).Artificial aeration increased TN concentration in the water column from 2011 to 2012; however, significant differences were not observed at any sampling locations. Furthermore, the historical mean TN concentration at the HMDI (2005-2008) is approximately 1.3 mg/L; the EPA recommended value is 0.44 mg/L (EPA, 2010). In comparison, the TN measured in this study was between 2 and 3 mg/L; therefore, the TN at the HMDI needs to be lowered.

In contrast, nitrate and nitrite concentrations showed significant increase at three sampling locations with aeration. Overall, aeration increased nitrate and nitrite concentration to 0.16 mg/L by the end of the sampling and that was greater than the historical concentration of nitrate and nitrite, which is, 0.03 mg/L at the HMDI between 1992 and 2008. Nevertheless, both nitrate and nitrite concentrations were less than the NDDoH recommended value of 0.25 mg/L (NDDoH, 2001).

In addition to nutrients, including N and P, aeration influenced DO and other factors. Compared to the non-aerated period, aeration raises the bottom DO concentration and causes the lake to become isothermal. Aeration successfully eliminated stratification and anoxic conditions

were not observed in 2010; however, in 2012, DO dropped when the aeration failed, indicating that turning off the aeration unit caused conditions to revert back to non-aeration values.

Results suggest that aeration benefits biological integrity by eliminating algae scums. The response, however, of total algal biomass to artificial aeration has been quite variable. Pastorok et al. (1982) reviewed 40 cases of complete destratification and found that the effect on chlorophyll *a* varied among lakes. Chlorophyll *a* concentration either increased, decreased or remained the same subject to the mixing conditions prevalent in individual lakes. The results of this study were consistent with reports of aeration suppression of cyanoacterial blooms (Jungo et al., 2001; Cowell et al., 1987). Current results showed that when dissolved inorganic N and P concentrations decreased at the surface layer during the non-aerated conditions, algal biomass also dropped rapidly. During the non-aerated conditions, the majority of bioavailable N and P was near the bottom layers and growth of phytoplankton was suppressed due to lack of dissolved nutrients at the surface water column, thus changes in chlorophyll *a* depended on limiting nutrients. In addition, these results indicate that there was no external loading of dissolved inorganic nutrients into the water column during the non-aerated period. If external loading of dissolved nutrients dominated during the non-aerated period in 2011, then algae population may not have been suppressed at the HMDI.

Overall, the mean chlorophyll *a* at the HMDI decreased significantly ($p<0.05$) with aeration in both years (2010 and 2012). Therefore, aeration suppressed total algal biomass by the end of the study. Based on lake water quality data, the dominant algae group at the HMDI was cyanobacteria; therefore, a possible explanation for suppression of algal biomass under aerated conditions could be a shift of the dominant algae group. For instance, a number of studies have documented the decline of abundance of cyanobacteria and an increase in abundance of

dinoflagellates and green algae with aeration in eutrophic lakes (Cooke et al., 2005; Burkholder, 2001). Favorable conditions for cyanobacterial blooms in lakes include calm stable water, high solar radiation and high water temperature (Havens et al., 2003). Since many filamentous cyanobacteria are positively buoyant under non-aerated conditions, the stable water column allows them to migrate to the surface to receive enough sunlight; however, with aeration, the entire water column is mixed, preventing these algae from remaining near the water surface resulting in less growth due to low irradiance (Havens et al., 2003). Other algal groups, such as dinoflagellates, become dominant with aeration because of their negatively buoyant capabilities, which are due to their heavy silica frustule (Reynolds, 1999). The Secchi depth, turbidity and TSS were highly varied and depended on the contribution of phytoplankton; however, conditions improved by 2012 under the conditions of aeration.

A limitation of the current study is the limited number of data representative of the non-aerated period, which are not statistically defensible. Therefore, expanding the number of sampling events for the non-aerated period will provide more statistically representative data to determine the influence of the aeration and the non-aeration on water quality.

CHAPTER 5. GENERAL CONCLUSIONS

5.1. Conclusions and Management Recommendations

Aeration is a viable means for lake associations to manage water quality in impaired lakes. Overall, this study revealed no significant differences in soluble reactive phosphorus (SRP), total phosphorus (TP) and total nitrogen (TN) concentrations in the water column between aeration and non-aeration conditions; however, those nutrient concentrations need to be lowered to the recommended North Dakota Department of Health (NDDoH) and the Environmental Protection Agency (EPA) standard levels. Because the Heinrich-Martin Dam Impoundment (HMDI) is a nitrogen-limiting, phosphorus-rich water body, lowering phosphorus (P) load rather than nitrogen (N) load is recommended. The mean ammonia-N concentrations at the near bottom layers during non-aerated conditions decreased significantly by artificial aeration. Therefore, to avoid summertime fish-kill due to ammonia toxicity, artificial aeration needs to be continued in the summer. In addition, the mean nitrate and nitrite concentrations increased significantly under aeration; however, the concentration did not exceed the NDDoH recommended value for lakes and reservoirs during the study period.

Another key finding was that aeration improved water quality as measured by Secchi depth, turbidity, dissolved oxygen (DO) and algal biomass. Further, this study indicated that aeration effectively eliminated undesirable features of eutrophication such as noxious algal blooms and oxygen depletion. Therefore, water quality can be greatly enhanced by operating the aerator in summer.

Since the HMDI is managed as a bluegill and largemouth bass fishery and as a recreational water body during both winter and summer, the current study results will provide further information to manage the HMDI more effectively under both aerated and non-aerated

conditions. This information will be useful for state agencies (NDDoH and NDG&F) to protect water resources and provide information to develop numeric water quality criteria for nutrients to protect the beneficial uses of the state's waters.

In addition, dissolved inorganic nutrients, TN, TP, temperature, and algae were homogenously distributed across the water column by aeration. The majority of phosphorous was dissolved inorganic phosphorous while the majority of N was dissolved organic nitrogen. Dissolved inorganic nutrients were distributed across depths under aerated conditions, which expanded the availability of nutrients for algal growth throughout the algal vegetative period. Therefore, an alternative aeration method is recommended to control nutrient redistribution by the existing aeration system. Further, a previous study (Overmoe, 2008) has shown that more than 90% of sediment at HMDI was in the form of organic sediments and total iron concentration in the water column was under the detectable level. Those results indicated that among possible dissolved inorganic phosphorus release mechanisms, such as Fe-desorption and organic matter decomposition, the dissolved phosphorus release by Fe-desorption was possibly minimal; nutrient release from organic matter decomposition would play a significant role. Therefore, alternative aeration methods for the HMDI would be hypolimnetic aeration or hypolimnetic oxygenation, which introduces fine air or oxygen bubbles that become dissolved in hypolimnetic water but do not disturb thermal stratification, which means that the eplimnion can be kept nutrient depleted.

In addition, current study results showed that under non-aerated conditions the majority of the TP was found in near bottom layers at deep locations; therefore, it is recommended that when TP concentration is maximum in the water column the existing hypolimnetic siphon be operated during lake stratification to remove the phosphorus - rich anoxic bottom water without

disturbing the thermal stratification. Further, in 2011, results indicated that lack of adequate dissolved inorganic nutrients from the hypolimnion to the epilimnion affected algal growth; therefore, hypolimnetic discharges of phosphorus - rich anoxic water suppresses algal growth by reducting of nutrient supply. Therefore, application of both in-lake measures, aeration and hypolimnetic discharges, would be beneficial in accelerating recovery at HMDI.

5.2. Recommendations for Future Work

In addition to issues addressed in this research, there are other topics that require further study. The sediment TN and TP concentrations and sediment organic matter concentration provide information to decide how much nutrient are trapped in the sediment to compare nutrient concentrations in the water column and in the sediment to find out the major source of nutrients at HMDI.

Further, a sediment core sample analysis under laboratory conditions will ensure nutrient release from lake sediments under aerated and non-aerated conditions. In addition, during precipitation events those nutrient concentrations will necessary to compare with nutrient concentrations in the water column in field conditions to confirm the major source of nutrients and contribution of nutrients from external sources versus internal sources in to the water column.

It will also be important to identify other cations and anions which affect water quality, such as carbonate and bicarbonate (for alkalinity measurements and total dissolved ions), and calcium, potassium, sodium, and chloride (for conductivity measurements and total ion concentrations). The major cations and anions concentrations in the water column could be used for modeling purposes to understand phosphorus precipitation/ fractions of available phosphates forms under different environmental conditions (aerated and non-aerated conditions).

Water quality parameters such as DO, water temperature, algal growth and nutrient concentrations are influenced by both seasonal variations and aeration. Therefore, to distinguish those influences, assessment of seasonal water quality under aerated and non-aerated conditions is recommended.

CHAPTER 6. REFERENCES

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